From a boreal bog to an abandoned peatland pasture: the effect of agricultural management and abandonment on the greenhouse gas fluxes, carbon balance and radiative forcing of a boreal bog in western Newfoundland, Canada

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Abstract

Undisturbed peatlands generally act as a long-term carbon (C) sink and climate cooling. Agriculturally managed peatlands have been identified as hotspots for C and greenhouse gases (GHGs) emissions. However, the increased magnitude of C and GHGs emissions following agriculture management was found to be significantly variable, dependent on the management intensity, peatland initial conditions, cultivation species, time for plant regeneration and fertilization amount. Moreover, the knowledge of how agricultural management and abandonment affects GHGs fluxes is limited by insufficient direct comparisons of GHGs fluxes between undisturbed peatlands and agriculturally managed ones and failure to consider all three GHGs species. To bridge this gap, I conducted a study measuring the landscape-scale carbon dioxide (CO_2) and methane (CH_4) fluxes by eddy covariance and plot-scale N_2O fluxes using static chamber technique in a boreal bog and an adjacent abandoned peatland pasture to determine and compare the controls on the temporal patterns of all GHGs fluxes and the effect of agricultural conversion and abandonment on the GHGs fluxes, the C balance and radiative forcing of a boreal bog in western Newfoundland, Canada.

This study showed that the gross primary productivity (GPP) and ecosystem respiration (ER) of the abandoned peatland pasture was significantly higher than the counterparts at the bog. The between-site difference in GPP was mainly related to their different vegetation conditions, and the between-site ER difference was linked to different conditions of water table, substrate availability and autotrophic respiration. Overall, the abandoned peatland pasture was a stronger CO_2 sink than the bog.

The abandoned peatland pasture was a smaller CH_4 source than the bog. CH_4 flux showed distinct diel and seasonal patterns at the bog but not at the abandoned peatland pasture. Subsurface soil temperature was the main control on CH_4 flux during the growing season but friction velocity became important in the non-growing season at the bog, while no variable was found to be significantly related to the seasonal variation in CH_4 flux at the abandoned peatland pasture.

N_2O flux did not show any significant temporal and spatial pattern and the fluxes were very low at both the bog and abandoned peatland pasture. The C and GHGs balance were mainly determined by the magnitude and direction of CO_2 at the pasture, but the GHGs balance was determined by CH_4 flux at the bog. The abandoned peatland pasture acted as a stronger C and GHGs sink than the bog. Therefore, results in this study suggest that the C sequestration capacity and climate cooling function of agriculturally managed peatlands can become stronger than the undisturbed peatlands after long-term abandonment.

Research in the thesis contributes new understanding of how agricultural management and abandonment affects the controls on GHGs fluxes and the C balance and climate regulation of peatlands. The work also shows the controls on C fluxes vary over different time-scales and different periods and pose difficulty for incorporation into ecosystem models.

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Contribution of authors

Chapter 3 to 6 are written in a format suitable for their publication in scientific journals. All chapters are co-authored by Jianghua Wu, Peter Lafleur, Junwei Luan, Huai Chen, Xinbiao Zhu. Jianghua Wu, as my supervisor, contributed to the planning of the research, giving helps in maintaining the eddy covariance towers and providing comments and edits to improve the manuscripts and offering financial support to the completion of the research. Peter Lafleur, as one of the academic committee members, provided constructive suggestions and edits to improve all the manuscripts. Junwei Luan helped install the eddy covariance towers and maintain the towers before I worked in the lab and helped collect the first year's N_2O gas samples. Huai Chen and Xinbiao Zhu gave some comments to improve the manuscripts.

Manuscript # 1 (Chapter 3) "Temporal shifts in controls over methane emissions from a boreal bog" by Mei Wang, Jianghua Wu, Peter Lafleur, Junwei Luan, Huai Chen, Xinbiao Zhu. (Agricultural and Forest Meteorology, in revision).

Manuscript # 2 (Chapter 4) "Near-zero methane emission from an abandoned boreal peatland pasture based on eddy covariance measurements" by Mei Wang, Jianghua Wu, Peter Lafleur, Junwei Luan, Huai Chen, Xinbiao Zhu. (Plos One, in revision).

Manuscript # 3 (Chapter 5) "Can the abandoned peatland pasture sequestrate more carbon dioxide than the adjacent boreal bog" by Mei Wang, Jianghua Wu, Peter Lafleur, Junwei Luan, Huai Chen, Xinbiao Zhu. (Agricultural and Forest Meteorology, accepted for publication).

Manuscript # 4 (Chapter 6) “Can an abandoned peatland pasture be a stronger greenhouse gas sink than an adjacent undisturbed boreal bog in western Newfoundland, Canada?” by Mei Wang, Jianghua Wu, Peter Lafleur, Junwei Luan, Huai Chen, Xinbiao Zhu. (To be submitted to GCB bioenergy for peer review).

Chapter 1 - Introduction

Peatlands are carbon (C) rich ecosystems and they currently represent nearly one-third of the global soil C pool, despite covering only 3% of the land surface (Gorham, 1991; Yu et al., 2010a). Undisturbed peatlands have acted as persistent C sinks, with the accumulation rate averaging ~15-30 g C m^{-2} yr^{-1} over thousands of years, mainly due to their low decomposition rates under the waterlogged and extremely cold soil conditions (Gorham, 1991; Turunen et al., 2002, 2004; Wang et al., 2014; Yu et al., 2010a). Cold, waterlogged conditions and *Sphagnum*-dominated recalcitrant litters in peatlands lead to low decomposition of peat organic matter (OM), benefiting C accumulation (Gorham, 1991). In this regard, the C sequestration capacity of peatlands is very sensitive to the alterations of temperature, hydrology and vegetation conditions, with the C balance easily challenged under the projected climate change and human activities (Martini et al., 2007). During the past century, approximately 20% of pristine peatlands have been converted for agricultural cultivation worldwide (Couwenberg, 2011; Joosten and Clarke, 2002; Turetsky and Louis, 2006). Such management involves drainage, crop/grass cultivation and fertilization (Turetsky and Louis, 2006), which can alter the peatland C and nitrogen (N) exchange, making agriculturally managed peatlands hotspots for C and other greenhouse gas (GHG) emissions (Grønlund et al., 2006, 2008; Kasimir-Klemedtsson et al., 1997; Lohila et al., 2004).

In Canada, peatlands cover an area of approximately 1.136 million km^2, second only to those in Russia (Tarnocai, 2005). Agricultural management of peatlands is the most common type of non-harvesting use in Canada (Joosten, 2009), with an area of 170,000 km^2 having been converted for

such use, accounting for 15% of the total national resource of peatlands and mires (Oleszczuk et al., 2008). However, limited studies have focused on the controls on C and GHGs flux of agriculturally managed peatlands and their difference with undisturbed ones, causing some bias in estimating and predicting how agricultural management affects the GHGs fluxes, C balance and radiative forcing in Canadian peatlands and their response to future changing climate and human activities.

1.1. Research questions and objectives

Agricultural management changes the hydrology and vegetation conditions in peatlands, which can cause significant impact on the dynamics of the GHGs fluxes and their role in global C cycle and climate regulation. This study measured carbon dioxide (CO_2) and methane (CH_4) fluxes by eddy covariance technique and nitrous oxide (N_2O) using static chambers to determine the biotic and abiotic controls on the C (CO_2, CH_4) fluxes across different timescales (diel, seasonal and inter-annual) and to quantify how agricultural conversion and abandonment would affect the C and GHGs balance of a boreal bog in western Newfoundland, Canada. The specific objectives are to 1) determine the diel, seasonal and inter-annual patterns of CH_4 flux and the shifting controls on the temporal variation, and quantify the annual CH_4 budget in the bog (chapter 3), 2) to determine the controls on the temporal variation in CH_4 flux and quantify the CH_4 budget in the abandoned peatland pasture (chapter 4), 3) to compare the CO_2 flux between the bog and the abandoned peatland pasture and determine the driving forces underlying the between-site difference in CO_2 flux (chapter 5), 4) to compare the full GHGs (CO_2, CH_4 and N_2O) and their radiative forcing effects between the bog and the abandoned peatland pasture (chapter 7). This study will provide insights for better understanding the changing controls on the dynamic patterns of the GHGs flux, C balance and radiative forcing between the undisturbed bog and

abandoned peatland pasture, help reducing the uncertainties to estimate the C balance and radiative forcing of agriculturally managed peatland and to quantify how agricultural management and abandonment affects the C balance and radiative forcing in boreal bogs and provide guidelines for making better and efficient policy-decisions with regard to land use and atmospheric GHGs.

1.2. Thesis outline

In addition to this introductory chapter (chapter 1), this thesis is composed of six main chapters. Chapter 2 synthesizes the current knowledge of the role of peatlands in the global C and N cycle and the controls on the processes as well as how these processes are affected by agricultural management. Chapter 3 to 6 are the main core of my thesis and each chapter presents one manuscript submitted for peer review. Chapter 3 presents controls on the diel and seasonal patterns of CH_4 flux and the annual budget of CH_4 flux at the bog, providing the insights of how the controls on CH_4 flux shift at different time-scales and different study periods at undisturbed boreal bogs. Chapter 4 presents the dynamic pattern of CH_4 flux and its controls and quantifies the annual budget at the abandoned peatland pasture, helping us better understand how CH_4 flux changes with the environmental variables and its annual budget at agriculturally managed peatlands. Chapter 5 compares the CO_2 flux between the bog and abandoned peatland pasture and identifies the variables explaining the between-site difference in CO_2 flux, providing the knowledge of the possible ways that agricultural management and abandonment impacts CO_2 flux. Chapter 6 compares all the GHGs fluxes, C balance and their radiative forcing between the bog and abandoned peatland pasture, giving insights on how agricultural management and abandonment affect the role of peatlands in global C cycle and their climate regulation function. Chapter 7 makes a conclusion about how the main findings in my thesis and how the findings

give insights on how C and GHGs changes under agricultural management and how this might influence their role in the global C cycle and climate regulation. I also identify knowledge gaps highlighted by my results.

Chapter 2 – Literature review on peatlands: carbon, greenhouse gases, agricultural management

2.1. The characteristics of peatlands: definition and distribution

Peatland ecosystems are a type of wetland where a thick layer of incompletely decomposed organic matter, or peat exists (Wieder et al., 2007). The required minimum thickness of peat can be found to be 40 cm (NWWG, 1988) and 30 cm (Laine and Vasander, 1996) in different definitions, making it difficult to estimate the area of pealtands on a global scale. Peatlands are categorized as bogs and fens according to the hydrologic connections with surrounding systems (Mitsch et al., 2009). Usually, the precipitation is the only nutrient and water source for bogs, resulting in nutrient-poor and acidic conditions, with pH less than 5 (Keddy, 2011). In contrast, fens can obtain water and nutrient from other systems and therefore they are nutrient-rich and alkaline ecosystems, with pH greater than 6 (Keddy, 2011). The main vegetation types in bogs are *Sphagnum* moss, sedge, ericaceous shrub or evergreen trees; fens are dominated by sedges and grasses (Keddy, 2011). The estimated area of global undisturbed peatlands is 400Mha, occupying an area of approximately 3% of the earth's land surface (Gorham, 1991; Joosten and Clark, 2002; Lappalainen, 1996; Maltby and Immirzi, 1993). Among them, an area of 346 Mha peatlands is distributed in boreal and subarctic regions, accounting for about 87% of the world's peatlands. The area of peatlands exceeds 5 Mha in six countries, where the peatland area is 142 Mha in Russia > 124 Mha in Canada > 62.5 Mha in USA > 27 Mha in Indonesia > 9.6 Mha in Finland > 7 Mha in Sweden (Joosten and Clark, 2002). Together the peatland area in these six countries accounts for ~93% of the total area of the world's peatlands.

2.2. The role of peatlands in global C cycle

2.2.1. Peat C pool

Though covering only 3% of the land surface, northern peatlands currently represent nearly one-third of the global soil C pool (Gorham, 1991; Yu et al., 2010a). Undisturbed peatlands have long been regarded as C sinks, mainly due to the imbalance between primary production and decomposition (Gorham, 1991). The estimated C pool of northern peatlands ranges from 234 to 547 Pg C (Armentano and Menges, 1986; Gorham, 1991; Lappalainen, 1996; Turunen et al., 2002; Yu et al., 2010a), between 1/3 and 3/4 that of the amount of atmospheric C (730 Pg C). Such large variations in estimated peat C pool are caused by different total area, bulk density and C content used in these calculations (Yu et al., 2010a).

2.2.2. CO_2 exchange in peatlands

The net ecosystem exchange of CO_2 (NEE) is determined by the processes of gross primary production (GPP) and ecosystem respiration (ER). In general, undisturbed northern peatlands have mostly acted as CO_2 sink, with the net uptake rate ranging from -2.5 to -144 g C m^{-2} yr^{-1} (Aurela et al., 2004, 2007; Bäckstrand et al., 2010; Christensen et al., 2012; Fortuniak et al., 2017; Lund et al., 2007, 2015; McVeigh et al., 2014; Nilsson et al., 2008; Olson et al., 2013; Peichl et al., 2014; Roulet et al., 2007; Strilesky and Humphreys, 2012; Syed et al., 2006). CO_2 fluxes of peatlands show obvious diel (Long et al., 2010), seasonal (Beetz et al., 2013; Bubier et al., 1998; Helfter et al., 2013; Hirano et al., 2007; Lafleur et al., 2001, 2003; Roulet et al., 2007; Syed et al., 2006; Wright et al., 2013) and inter-annual patterns (Euskirchen et al., 2014). Strong NEE diel patterns in peatlands are typically observed during summer season, with net CO_2 uptake occurring in the daytime and emission in the nighttime, driven by the diel pattern of plant photosynthesis (Lafleur et al., 2001; Zhou et al., 2009). The CO_2 flux seasonal pattern shows that

peatlands absorb CO_2 from the atmosphere during the growing season, with the NEE peak in mid-growing season, while they act as a CO_2 source in winter (Beetz et al., 2013; Bubier et al., 1998; Helfter et al., 2013; Hirano et al., 2007; Lafleur et al., 2001, 2003; Roulet et al., 2007a; Syed et al., 2006; Wright et al., 2013), in line with the seasonality of temperature, soil moisture, hydrology and vegetation phenology (Hirano et al., 2007; Syed et al., 2006; Wright et al., 2013). Inter-annual variation in NEE is mainly driven by alterations in growing season length, temperature, soil moisture and water table position (Lafleur et al., 2003; Peichl et al., 2014; Teklemariam et al., 2010; Webster et al., 2013).

2.3. GPP and its controls

GPP, which determines the inputs of new plant organic matter into the ecosystem, is arguably important ecological function of peatlands. There are several ways to estimate primary production, including direct ground measurements of plant production (Bartsch and Moore, 1985; Camill et al., 2001; Hirota et al., 2007; Kosykh et al., 2008; Laiho et al., 2014; Moore et al., 2002; Murphy and Moore, 2010), C fluxes (Dimitrov et al., 2011; Grant et al., 2012) or ecosystem modelling (Dimitrov et al., 2011; Harris and Dash, 2011). Ground measurements often involve destructive sampling techniques. The total aboveground production varied significantly, ranging from 78 to 1045 g m^{-2} (Bartsch and Moore, 1985; Dimitrov et al., 2011; Moore et al., 2002; Murphy and Moore, 2010; Vasander, 1982; Wieder et al., 1989). The GPP based on eddy covariance measurements averaged 514 ± 206 g C m^{-2} yr^{-1} for 12 northern peatland and tundra sites (Lund et al., 2010), lower than their grassland counterparts (most above 800 g C m^{-2} yr^{-1}) (Ma et al., 2007; Nagy et al., 2007; Xu and Baldocchi, 2004). GPP in northern peatlands is mainly controlled by the light availability, hydrologic and thermal conditions, availability of nitrogen (N) and phosphorus (P), plant phenology and composition (Helfter et al., 2015; Lund et

al., 2009; Nijp et al., 2015; Sulman et al., 2010).

2.3.1. GPP and light availability

Plant photosynthesis is controlled by light availability that provides energy for the C assimilation process. Many previous studies suggest a strong and positive relationship between GPP and PPFD in a wide range of ecosystems including peatlands (Lee et al., 2015; Nijp et al., 2015; Schubert et al., 2008, 2010). Therefore, PPFD/PAR is considered as an important variable modeling GPP (Schubert et al., 2008, 2010).

2.3.2. GPP and nutrient availability

Peatlands are nutrient limited ecosystems and the growth of plants is usually limited by nutrient availability. High leaf concentrations of both N and P enhance photosynthesis (Baddeley et al., 1994; Leith et al., 1999). N controls the amount of ribulose-1, 5-bisphosphate carboxylase oxygenase (Rubisco), which is the enzyme that fixes C in C_3 plants (Baddeley et al., 1994; Farquhar et al., 1980). P limits the carboxylation rate at low concentrations (Harley et al., 1992). It should therefore be expected that photosynthesis rate and plant growth are affected by N and P availability. Field experiments show that N addition significantly increased GPP in bogs subjected to low N deposition, while GPP was stimulated by P addition in the high N deposition bogs in Sweden (Lund et al., 2009) and the Netherland (Aerts et al., 1995). Moreover, different plants in peatlands respond differently to N addition, with N amendment exerting a negative effect on the growth of *Sphagnum* moss but a positive influence on the growth of vascular species (Berendse et al., 2001). Berendse et al. (2001) reported that increased N deposition reduced *Sphagnum* moss growth, because it increased the cover of vascular plants and the tall moss *Polytrichum strictum* that caused shade for *Sphagnum* moss growth at a bog in Sweden.

Bubier et al. (2007) found the highest levels of nutrient addition showed lower rates of maximum NEE and GPP after 5 years in a fertilization experiment at Mer Bleue Bog, Canada. A meta-analysis by Limpens et al. (2011) suggested that high N loading depressed *Sphagnum* production relative to untreated control, due to the interactive effects of presence of competitive vascular plants and high tissue N concentrations that altered nutrient stoichiometry.

2.3.3. GPP and soil moisture

Soil moisture has been identified as an important variable regulating plant growth. Several studies have documented reduced GPP in peatlands during drought events due to the effects of moisture stress on plant productivity (Alm et al.,1999; Chivers et al., 2009; Weltzin et al., 2000). Moreover, Lund et al. (2012) suggested that GPP was most sensitive to drought during the period of leaf out and canopy development compared with the full canopy stage. Sulman et al. (2010) found different responses of GPP to drought between fens and bogs, with wetter conditions reducing GPP at fens but enhancing GPP at bogs, due to differences in the relative contributions of vascular plants and mosses. However, other studies have found that lowered water table can stimulate GPP by promoting vegetation productivity and/or stimulating nutrient availability for plant uptake (Gorham 1991; Munir and Strack, 2014; Thormann et al., 2001; Weltzin et al., 2000). Munir and Strack (2014) found that the longer-term drainage resulted in the replacement of mosses with vascular plants (shrubs) on the hummocks leading to increase in CO_2 uptake in a boreal treed bog.

2.3.4. GPP and plant phenology

Plant phenology has shown to be an important regulator in CO_2 flux in peatlands (Helfter et al., 2015; Kross et al., 2014; Lund et al., 2009). The onset of GPP in each year was closely related to

the thinning and disappearance of the snow cover, initially associated with moss activity (*Sphagnum* and *Polytrichum*), followed by the evergreen shrubs (Moore et al., 2006). Kross et al. (2014) found that peatlands with earlier growing seasons had the higher average annual GPP. In addition, growing season length is important in determining the spatial and temporal variation in GPP (Helfter et al., 2015).

2.4. ER and its controls

The decomposition rate of organic matter in peatlands is very slow under cold and anoxic conditions (Gorham, 1991). Indeed, previous studies suggested that the decomposition rates of litter in peatlands was low, with the exponential decay values ranging from -0.03 to -0.3 yr^{-1} of fresh plants and from -0.02 to -0.0001 yr^{-1} of organic matter stored in peat soils (Clarkson et al., 2014; Moore and Basiliko, 2006). Moreover, the decomposition rate in surface soil was generally higher than that in subsurface soils, decreasing from -0.24 yr^{-1} to -0.04 yr^{-1} (Clarkson et al., 2014) and from -0.05 yr^{-1} to -(0.002-0.02) yr^{-1} (Scanlon and Moore, 2000).

Many factors regulate litter decomposition, e.g. litter quality, temperature, moisture, the oxygen associated with water table (WT) and the microbial activity (Clarkson et al., 2014; Moore and Basiliko, 2006; Scanlon and Moore, 2000; Ward et al., 2013).

2.4.1. ER and litter quality

Litter quality is an important factor affecting decomposition rate, causing it to vary among different species and components of species (Belyea, 1996; Krab et al., 2013; Strakova et al., 2012; Thormann et al., 2001; Turetsky, 2004; Ward et al., 2013). Generally, the decomposition rate from the fastest to the slowest is ranked as follows: sedge > shrub leaves > tree leaves > tree

needles > hollow/lawn Sphagnum > hummock Sphagnum > tree woody material. The difference in the decomposition rate of different litter species is mainly regulated by different litter chemistry conditions. It was found that the ratio of lignin to N of the initial litter contributed mostly to the decomposition variations of different species (Moore and Basiliko, 2006). Sphagnum, with complex organic compounds and low nutrient content， decomposes very slowly (Moore and Basiliko, 2006; Moore et al., 2007).

2.4.2. ER and soil temperature

Soil temperature, which regulates microbial activity for mineralization has been suggested as a dominant variable in regulating soil respiration (Aljaste, 2011; Cai et al., 2010; Chivers et al., 2009; Lafleur et al., 2005a). Many field studies found a strong correlation between soil temperature and ER in peatlands (Aljaste, 2011; Cai et al., 2010; Chivers et al., 2009; Lafleur et al., 2005a). Usually, CO_2 production rates increase by a factor of 2-3 for every 10^{o}C temperature ranges (Q_{10}) (Bubier et al., 2003; Moore and Dalva, 1993a). Lower Q_{10} values ranging from 1.2 to 1.6 have been found in several peatlands in Canada (Yavitt et al., 2000) and higher values of up to 8.7 in peatlands in Poland (Szafranek-Nakonieczna et al., 2014). Moreover, Q_{10} varies significantly for peats at different depths. For example, Szafranek-Nakonieczna et al. (2014) found that Q_{10} values decreased from 8.72 in peats at 0-20 cm to 4.15 in peats at 60-80 cm in an eastern Poland peatland.

2.4.3. ER and soil moisture/water table

Soil moisture / WT is another important regulator for ER of peatlands since it is related to the oxygen availability in the uppper peats (Chimner and Cooper, 2003). Water table drawdown exposes a large amount of peat to aerobic conditions and enhances their mineralization. Many

studies previously suggested that drainage or lowering WT signifcantly increased the ecosystem respiration (Chivers et al., 2009; Salm et al., 2012; Strack et al., 2006), though the increase in ER following WT drawdown may be limited by labile substrate availability in subsurface peats. Chimner and Cooper (2003) found large increase in CO_2 emissions when lowering the WT beneath the soil surface, but no increase in emissions occurred as they furthered lowered water tables beneath the soil surface. However, weak correlation between ER and WT was observed at Mer Bleue bog, probably due to compensations between production of CO_2 in the upper and lower peat profile as the WT falls and the importance of autotrophic respiration, which is relatively independent of water table position (Lafleur et al., 2005a).

2.4.4. ER and soil nutrient availability

Nutrient addition has been shown to be strongly linked with C cycling processes in peatlands. N addition favors the growth of vascular plants but inhibits *Sphagnum moss's* (Davidson and Janssens, 2006; Moore and Dalva, 1993). This species change with more vascular plants and less *Sphagnum* moss provides a large amount of faster decomposing litters and fuels the C and N cycling and therefore loss of CO_2 from peatlands. In addition, the loss of *Sphagnum* moss may result in drier and perhaps warmer soils with the loss of moss water holding capacity, which may lead to an increase in aerobic microbial decomposition and CO_2 emission (Davidson and Janssens, 2006; Moore and Dalva, 1993). Indeed, many field studies have suggested that ER increases following N amendment in peatlands without other nutrients limitation (Amador and Johnes, 1993; Juutinen et al., 2010; Kivimäki et al., 2013). Moreover, different N forms can cause different impacts on peatland ER, with adding nitrate increased the CO_2 loss more than ammonium, both with and without P and potassium (Kivimäki et al., 2013).

2.5. CH_4 flux and its controls in peatlands

Peatlands are important natural CH_4 sources, with an average annual emission of 70 Tg yr^{-1} (Gorham, 1991). The CH_4 emission rate ranges from 3.7 to 65.7 g C m^{-2} yr^{-1}, varying among different peatlands (Moore et al., 2011; Treat et al., 2007; Turetsky et al., 2008) as well as micro-topographies within a peatland (Munir and Strack, 2014; Strack et al., 2004). Managed peatlands are typically a weak CH_4 source or even a small sink, with the CH_4 flux ranging from -0.03 to 1.6 g C m^{-2} yr^{-1} (Grønlund et al., 2006; Kasimir Klemedtsson et al., 2009; Kløve et al., 2010; Maljanen et al., 2001, 2010; Regina et al., 2007). CH_4 fluxes show obvious seasonal and inter-annual variation patterns, primarily due to the dynamics of temperature and WT position (Dinsmore et al., 2010; Jackowicz-Korczyński et al., 2010; Moore et al., 2011; Olson et al., 2013; Treat et al., 2007).

The CH_4 flux of peatland is determined by the processes of CH_4 production, consumption and transport, which can be affected by many physical, chemical and biological factors (Lai, 2009).

2.5.1. CH_4 flux and water table

WT is one of the key factors in regulating the CH_4 flux in peatlands since it determines the thickness of CH_4 production and consumption zones. WT is typically considered to be positively correlated with CH_4 flux and lowering WT can reduce CH_4 emissions (Munir and Strack, 2014; Strack et al., 2004; Yrjäläet al., 2011). The underlying mechanism is that WT drawdown is associated with a decrease in the thickness of CH_4 production zone and an increase in the thickness of CH_4 oxidation zone, resulting in less CH_4 release (Lai, 2009). However, the relationship between WT and CH_4 flux can be complicated under field conditions. Treat et al. (2007) reported an increase in CH_4 emission from a temperate fen as WT dropped over time,

which might be due to the overriding effects of increased CH_4 production and ebullition arising from higher peat temperature and reduced hydrostatic pressure on CH_4 flux. Brown et al. (2014) found that peak CH_4 emission at the Mer Bleue bog occurred when WT was 40-55 cm below the peat surface, instead of being close to the surface. They speculate this critical zone coincides with the necessary redox potentials and sources of fresh organic material that lead to maximum production of CH_4 and/or with conditions that lead to degassing of stored CH_4. Moreover, the relationship between WT and CH_4 emission derived from correlation analyses is further complicated by lagged flux response to varying WT (Kettunen et al., 1996) and strong hysteresis between CH_4 fluxes on the falling and rising WT limbs (Brown et al., 2014; Moore and Roulet, 1993). Moreover, some studies have suggested that WT acted as an "on-off" switch in regulating CH_4 flux: it was the dominant variable driving CH_4 flux when water table dropped below a certain threshold, but other factors became more important when water table was raised above that threshold (Brown et al., 2014; Luan and Wu, 2015).

2.5.2. CH_4 flux and soil temperature

Temperature is another variable closely related to CH_4 flux in peatlands. CH_4 flux is generally positively related to temperature (Long et al., 2010; Pypker et al., 2013; Rinne et al., 2007). An incubation experiment suggested CH_4 emission was much higher at a higher temperature (Treat et al., 2014). Although the temperature increase can enhance methane production as well as CH_4 oxidation, the increased CH_4 production outweighed the increase in CH_4 consumption (van Winden et al., 2012). Temperature was shown to be a significant control on the temporal variation in CH_4 flux, with high emissions occurring in warm period (Long et al., 2010; Pypker et al., 2013; Rinne et al., 2007). In addition, warming treatment was found to increase CH_4 emission in Alaskan wetlands (Turetsky et al., 2008) and in both bogs and fens in northern

Minnesota, USA (Updegraff et al., 2001; White et al., 2008). However, the response of CH_4 flux to warming can be modulated by micro-topography (Johnson et al., 2013; Munir and Strack, 2014), hydrology (Yang et al., 2014) and vegetation conditions (Ward et al., 2013). This modulation has led to either no or even negative effect of warming on CH_4 release in a poor fen (Johnson et al., 2013) and a boreal bog (Munir and Strack, 2014). Granberg et al. (2001) found that increasing temperature impacted CH_4 emissions positively when sedge cover was high, but showed no effect when sedge cover was low. Moreover, temperature plays a dominant role in regulating CH_4 flux when WT was high but loses its importance in governing CH_4 flux when WT is very low (Luan and Wu, 2015).

2.5.3. CH_4 flux and peat acidity

Peat acidity can regulate CH_4 flux by regulating methanogenic and methanotrophic activities. Although some acidophilic methanogens like *Methanobacterium espanolae* were found to grow at pH as low as 5.5 (Dunfield et al., 1993; Kamal and Varma, 2008; Kotsyurbenko et al., 2004), the majority of methanogenic bacteria grow only under a narrow pH range between 6 and 8 (Garcia et al., 2000). Valentine et al. (1994) observed a significant reduction in CH_4 production potential of northern fen surface peat by approximately sevenfold by lowering the pH of peat slurry from 7 to 5.5. Low pH inhibits CH_4 production through both direct inhibition of methanogenesis pathways and indirectly through its effects on fermentation, but the direct effects are stronger in ombrotrophic peatlands (Ye et al., 2012). The growth of methanotrophs is favored at more acidic pH ranging from 4.3 to 5.9 (Kamal and Varma, 2008), though there is evidence of methanotrophic bacterial growth at pH below 1 (Pol et al., 2007). However, these optimal pH condition of methanogenesis and methanotrophs are seldom reached in the field, especially in *Sphagnum*-dominated ombrotrophic bogs where the pH of peat water typically lies between 3

and 4.5 (Dedysh et al., 2002; Dunfield et al., 1993). Dunfield et al. (1993) suggested that the microflora involved in CH_4 metabolism is not well adapted to low pH values, with the optimum pH values which were about 2 pH units higher than the native peat pH values and only 0-1 pH unit higher than in more alkaline peats.

2.5.4. CH_4 flux and vegetation

Vegetation is another key factor affecting CH_4 flux of peatlands. It was suggested that changes in the plant species composition may alter the important processes relating to controls of and interactions between CH_4 fluxes from peatlands (Ström et al., 2005). Dias et al. (2010) even suggested the use of plant species composition as a simple and effective alternative for deriving predictions of CH_4 emissions in peatlands in the context of land use change. Aerenchymatous plants, with deep root systems that penetrate into the anoxic zone, can promote the production through root exudation and transport of CH_4 via providing gas conduits (Joabsson et al., 1999; Moore et al., 2011). Many studies have shown that CH_4 flux is closely linked to the presence of aerenchymous vegetation, with higher CH_4 released from areas containing aerenchymatous plants than those without such vegetation (Waddington et al., 1996a; Yrjäläet al., 2011). However, Strack et al. (2006) suggested different functions of sedges on the CH_4 flux under different water conditions where sedges could further attenuate CH_4 fluxes during dry periods but cause more CH_4 emissions during wet periods. Moreover, the plant productivity also controls the CH_4 flux by regulating the substrate availability for CH_4 production and much evidence suggested a positive correlation between CH_4 flux and CO_2 uptake (King et al., 1998; Waddington et al., 1996a). However, Waddington et al. (1996b) indicated that the coupling correlation between CH_4 and net ecosystem exchange only existed in wet sites or sites dominated by vascular plants. Lai et al. (2014) further suggested that the coupling of plant production and

CH_4 emissions is dependent on species composition, WT, and plant phenology and found a lag of CH_4 emission behind plant productivity of 9-12h. Pypker et al. (2013) found that the daily CH_4 flux increased with CO_2 uptake rate when daily NEE was negative, but no correlation was observed in days when little CO_2 was fixed.

2.5.5. CH_4 flux and nutrients availability

CH_4 flux can be also impacted by the availability of nutrient, trace metals, sulfur (S) and other chemicals. Peatlands are nutrient limited ecosystems and addition of nutrients like glucose, ethanol and acetate and H_2/CO_2 were previously reported to increase CH_4 production (Coles and Yavitt, 2002; Yavitt and Seidman-Zager, 2006). Moreover, the quality of substrate can influence the rate of methanogenesis. An increase in cellulose content and decrease in lignin to nitrogen ratio in peat are reported to increase peat decomposability and hence CH_4 production potential in northern peatlands (Lai, 2009). In addition, Basiliko and Yavitt (2001) showed that the addition of trace metals (Fe, Ni, and Co) and sodium significantly enhanced the production rate of CH_4 from *Sphagnum*-dominated North American peatlands by increasing the metabolism and growth of methanogens. However, addition of S had negative impact on CH_4 flux (Granberg et al., 2001). SO_4^{2-} from acid rain was suggested to inhibit the CH_4 emission since sulfate-reducing bacteria outcompeting methanogens for substrates in peatlands (Gauci et al., 2002; Nedwell and Watson, 1995).

2.6. N_2O flux and its controls in the peatlands

N_2O with a warming potential 295 times that of CO_2 on a 100 years horizon is a potent greenhouse gas whose concentration in the atmosphere has increased since the Industrial Revolution at an average rate of 0.77 ± 0.03 ppb yr^{-1} and which contributes nearly 10% of the

total anthropogenic radiative forcing (IPCC, 2014). In addition, N_2O is among the key chemicals destroying the stratospheric ozone layer (Ravishankara et al., 2009). Therefore, there is a strong interest to expand our knowledge on the mechanisms contributing to N_2O production and consumption in various ecosystems and how these processes are regulated by environmental factors.

N_2O is mainly produced through microbial processes of nitrification and denitrification (Koponen et al., 2006; Mosier et al., 2004; Regina et al., 1996), though other sources exist in the processes of dissimilatory nitrate reduction to ammonium and nitrifier denitrification (Hu et al. 2015; Li and Lang, 2014; Wrage et al. 2005). Nitrification is the process of oxidizing ammonium (NH_4^+) or ammonia (NH_3) to nitrate (NO_3) by nitrifying bacteria (Bollman and Conrad, 1998; McLain and Martens, 2005) and N_2O is released as an intermediate product mainly under aerobic conditions (Dalal et al., 2003). Denitrification is an anoxic process where denitrifying bacteria reduce NO_3- to N_2 and along this reduction gradient N_2O and NO are also produced and partly emitted to the atmosphere (Anderson and Poth, 1989; Baggs and Philipot, 2010). Denitrification is considered as the major contribution to the N_2O production in wetlands due to anaerobic conditions.

In pristine boreal peatlands, N_2O fluxes are typically very low in ombrotrophic (nutrient-poor) bogs and often below the detection limit, while in minerotrophic (nutrient-rich) fens emissions are larger (Regina et al., 1996; Song et al., 2009). Moreover, N_2O emission hotspots were found in peat-circles of the tundra (Repo et al., 2009) with the high emissions being explained by the favorable conditions for N_2O production, such as the O_2 availability enabling nitrification, the

low C/N-ratio and the availability of labile C (Klemedtsson et al., 2005). In addition, high N_2O flux was also found in the cold winter time, probably due to many cycles of freezing and thawing in the winter (Maljanen et al., 2009; Regina et al., 2004).

N_2O production is affected by many physical and biochemical factors, such as nitrate and O_2 concentration, organic matter content, temperature, soil pH and soil moisture content (Horn, 1994; Yu et al. 2001).

2.6.1. N_2O flux and soil temperature

Nitrification and denitrification rates increase with temperature within a certain range of 25-35 ℃ (Haynes and Sherlock, 1986; Koponen et al., 2006). It was found that N_2O emissions enhanced with soil temperature increasing from 10 to 15 ℃ (Lang et al., 2011). Allen et al. (2010) showed that N_2O emissions were higher during wet and hot months than in cool and dry months. However, elevated temperature affects multiple processes, some of which may offset N_2O emissions and result in an overall small net effect. Firstly, temperature can promote the growth of plants and N uptake, which are a competitor for NO_3^- with denitrifying microbes. Silvan et al. (2005) found *Eriophorum vaginatum* L., a successful competitor for NO_3^- with denitrifying microbes, and consequently moderated the N_2O emissions from peatlands. Secondly, warm climate may cause soil drying, which would have negative effect on N_2O flux (Bijoor et al., 2008; McHale et al., 1998). Thirdly, warming often has no effect on, or sometimes even decreases, belowground C input (Dieleman et al., 2012). If N_2O is mainly produced by denitrifiers that are C-limited, then warming conditions would have little effect on N_2O flux. Lastly, in field experiments soil, air, or canopy temperatures were increased by 1-5 ℃ and the increase magnitude in temperature was relatively small, causing a minor effect on N_2O. Therefore, soil

temperature can exert a positive, no or even negative effect, on N_2O emission in the field (Lohila et al., 2010; Yu et al., 2010b).

2.6.2. N_2O flux and soil moisture

Soil water content not only determines the availability of O_2, but also affects transport of nutrients within the soil matrix and the metabolic activity of microbial cells (Hu et al., 2014), and acts as an essential factor regulating N_2O production and the contribution of nitrification and denitrification to N_2O emission from soils (Cheng et al., 2014; Hu et al., 2015). Generally, the rate of N_2O production is low below 40-70% water-filled pore space (WFPS), but increases rapidly when WFPS exceeds this threshold (Bateman and Baggs, 2005; Dalal et al., 2003; Mathieu et al., 2006; Takakai et al., 2006). Pihlatie et al. (2004) suggested that the highest N_2O production, measured at the wettest soils (100% WFPS), was up to four orders of magnitude higher than that at the dry soils (40% WFPS). However, a high soil water content with WFPS above 75% was suggested to stimulate the reduction of N_2O to N_2 in denitrification, thus inhibits the production of N_2O (Davidson, 1992; Saggar et al., 2013; Weier et al., 1993).

2.6.3. N_2O flux and water table

N_2O emissions from waterlogged peat soils are usually negligible (Augustin et al., 1998; Martikainen et al., 1993; Regina et al., 1996). WT drawdown has been found to increase N mineralization in peat (Updegraff et al., 1995), and to enhance N_2O fluxes in both field and laboratory experiments (Freeman et al., 1992; Kliewer and Gilliam 1995; Martikainen et al., 1993; Regina et al., 1996). However, Martikainen et al. (1993) found that lowering WT had no effect on N_2O emissions from nutrient-poor peat but enhanced those from nutrient-rich peat and the highest N_2O emissions are found at intermediate groundwater levels, which allow their

aerobic and anaerobic production.

2.6.4. N_2O flux and N addition

Peatlands are nutrient-limited ecosystems and N_2O flux is usually limited by N availability. Previous studies suggested that N_2O flux can be enhanced by adding N (Flessa et al.,1995; Regina et al, 1998; Smith et al., 1998) and the weather after N fertilization also affects the magnitude of N_2O flux increase (Flessa et al., 1995; Smith et al., 1998). Augustin et al. (1998) suggested that N_2O emission was only stimulated when N application rates were very high (480 kg N ha^{-1}), and did not increased under a moderate N supply (60 or 120 kg N ha^{-1}). Aerts and Ludwig (1997) found that the nutritional status of the peat had a profound influence on the effect of water table lowering on N_2O emission: N_2O emission from the eutrophic soil was strongly increased at a lower water table, whereas there was no detectable N_2O emission from the mesotrophic soil.

2.7. The effect of agricultural management on GHGs of peatlands

During the last two centuries, the area and intensity of agricultural production on peatlands have increased in many countries and agriculture is now the most widespread human use for peatlands globally (Joosten and Clarke, 2002). Since the Second World War, large areas of peatland in central and Eastern Europe have been converted into pastures, hay meadows, ploughed fields, forestry plantations and fishponds (Joosten and Clarke, 2002). About 14% of European peatlands are currently used for agriculture, the great majority being used as meadows and pastures (Ilnicki, 2002; Joosten and Clarke, 2002; Lappalainen, 1996) and only a few percent as arable land (Joosten and Clarke, 2002). For example, in the Netherlands, about 43,782 ha are currently used as arable land and 178,226 ha are under grass. In countries such as the Netherlands (85%),

Germany (85%) and Poland (70%), almost all organic soils are cultivated. Elsewhere, only small areas of peatland are currently under agricultural use (Finland, 2%; United Kingdom, 4%; Sweden, 5%). In the United States the largest areas of peatlands are situated in Alaska, Minnesota, Michigan, Florida and Wisconsin. Over 230,000 hectares of fen peatland in the Florida Everglades are cultivated mostly for sugar cane and rice. Only 20% of Indonesia's peatlands are drained and agriculturally used (Ilnicki, 2002; Joosten and Clarke, 2002). Canada has one of the largest areas of agriculturally used peatland (170000 km^2) but this amounts to only 15% of the total national resource of peatlands and mires (Table 2.1).

2.7.1. Agricultural management and hydrological characteristics

Agricultural management accompanied by water table drawdown directly influences the hydrological conditions of peatlands. The responses of hydrological characteristics to drainage in peatlands have long been studied (Holden et al., 2004). Water table drawdown leads to the collapses of surface peats and destroys macropores (Silins and Rothwell, 1998), which are important pathways for runoff (Baird, 1997; Holden et al., 2003). As a result, the bulk density increased but hydraulic conductivity decreased (Kopp et al., 2013; Whittington and Price, 2006). When compared to undisturbed peatlands, drained peatlands were found to be more sensitive to rainfall: the WT was lower and more fluctuating (Holden et al., 2011; Kopp et al., 2013), with increased peak flows (Robinson, 1986; Skaggs et al., 1994) and shorter lag times (Conway and Millar, 1960). Drier condition in drained peatlands can increase the capillary action, causing more water loss from subsurface peat layers and thus the dryness and shrinkage of more peats (Hobbs, 1986). Once peat dries, it often becomes hydrophobic and cannot regain its initial moisture content (Eggelsmann et al., 1993). Permanent structural change of peats was found under drought simulation in the laboratory (Holden and Burt, 2003), causing alterations of the

hydrological routing of water through the peat tested.

Table 2.1. The area of peatland for agricultural use in some countries of the world (Ilnicki, 2002; Joosten and Clarke, 2002; Lappalainen, 1996; Mutalib et al., 1991).

Region	Country	Total peatland area (km^2)	Peatland drained for agriculture area (km^2)	Peatland drained for agriculture percentage (%)
North America	Canada	1114000	170000	15
	USA	611000	61000	10
Europe	Belarus	23967	9631	40
	Estonia	10091	1300	13
	Finland	94000	2000	2
	Germany	14200	12000	85
	Great Britain	17549	720	4
	Iceland	10000	1300	13
	Ireland	11757	896	8
	Latvia	6691	1000	15
	Lithuania	4826	1900	39
	Netherlands	20350	2000	85
	Norway	23700	1905	8
	Poland	10877	7620	70
	Russia	568000	70400	12
	Sweden	66680	3000	5
	Ukraine	10081	5000	50
Asia	Indonesia	200728	42000	20
	Malaysia	25890	8285	32
	China	10440	2610	25

2.7.2. Agricultural management and nutrient

Lowering water table level exposed upper peats into aerobic conditions, which in turn affect

microbial processes and thus decomposition rate and nutrient release. N mineralization increased strongly with water table drawdown (Venterink et al., 2009). P and K limitation was found in a drained Polish fen due to increased nutrient release (Mars et al., 1996). Indeed, previous studies suggested that agricultural drainage caused sediment and nutrient losses, especially for nitrate-nitrogen and soluble salts (Miller et al., 1996; Skaggs et al., 1994). Similarly, drained catchment experienced the highest nutrient leaching (Kløve et al., 2010). It was also found that the concentrations of dissolved organic nitrogen and ammonium (Laine et al., 2013), NH_4-N and silica (Miller et al., 1996), dissolved organic carbon (DOC) (Holden et al., 2004) increased after artificial water table drawdown.

2.7.3. Agricultural management and vegetation

Agricultural management is always associated with plant cultivation, grass or crops, resulting in increased presence of vascular plants on the managed peatlands (Oleszczuk et al., 2008). In addition, agricultural management is always followed by vegetation shifts since different species respond differently to lowering water depth. Species which are tolerant of drier conditions dominate drained peatlands, and those that are restricted to moist or saturated soils lose their dominance or even disappear in drier condition (Fisher et al., 1996). Drainage was found to decrease the coverage of *Sphagnum* moss on hummocks and promoted the growth of sedges on lawns in a boreal peatland in Canada (Strack et al., 2006) and Sweden (Breeuwer et al., 2009). Moreover, these vegetation shifts may differ among different microforms, with the coverage of *Sphagnum* declined on drained hummock, while Sphagnum dominated the drained hollows (Strack et al., 2006). In addition, water table can also affect the biomass and biomass allocation of peatlands, with water table drawdown enhancing and decreasing the above-ground and below-ground biomass, respectively, and promoting the influx of labile carbon to peat soils via fine

roots and leaves of a boreal peatland at Mer Bleue (Murphy and Moore, 2010).

2.7.4. Agricultural management and CO_2 flux

CO_2 emissions from soils arise from respiration by plant roots and living organisms, and from the mineralization of organic matter, dead micro-organisms, soil animals and dead plants, which is largely related to the availability of oxygen and OM (Marscher and Bredov, 2002; Marschher and Noble, 2000). The drainage for agricultural use increases subsurface oxygen and the degradation of dissolved organic matter available and induces large CO_2 emissions from managed peatlands. It was found that drained peatlands subsided at an average rate of 19 ± 2 mm yr^{-1} during 2000-2012 and 26 ± 1 mm yr^{-1} during 1920-2000 in New Zealand (Pronger et al., 2014). About 70-80% of the long-term subsidence of drained agricultural peat soils is caused by peat oxidation and mineralization (Ilnicki, 2002; Okruszko, 1993). Review works suggest that drainage for agricultural management has turned peatlands into significant CO_2 source in Nordic countries (Maljanen et al., 2010), UK (Ostle et al., 2009) and Canada (Oleszczuk et al., 2008). The CO_2 loss from agricultural peatlands in Nordic countries averaged 633 g C m^{-2} yr^{-1} (Maljanen et al., 2010), higher than that of 91 g C m^{-2} yr^{-1} in Canada (Oleszczuk et al., 2008), but lower than that of 1118 g C m^{-2} yr^{-1} for those in temperate regions. Moreover, CO_2 emission varied significantly among different agricultural managements, with larger CO_2 emission of 782 g C m^{-2} yr^{-1} from sites drained for grass and 683 g C m^{-2} yr^{-1} from agricultural soils without vegetation than that of 479 g C m^{-2} yr^{-1} from sites drained for barely and 684 g C m^{-2} yr^{-1} from abandoned sites with vegetation (Maljanen et al., 2010). In addition, studies from individual sites indicates that the agriculturally managed peatlands were CO_2 sources, with the emission rate of 548 g C m^{-2} yr^{-1} in an Atlantic peat bog in northwest Germany (Beetz et al., 2013) and 341 g C m^{-2} yr^{-1} in San Francisco Delta, USA (Knox et al., 2015). A comparison study suggested that

CO_2 emissions are three times greater in a drained peatland than that in a pristine one in Quebec (Waddington and Price, 2000). Peat mineralization and CO_2 emission are most intense when the water table is 90 cm below the soil surface. Lowering of the water table from 50 cm to 90 cm depth increased the annual CO_2 emission from 10,000 kg ha^{-1} yr^{-1} to 30,000 kg ha^{-1} yr^{-1} in the Netherlands (Wösten and Ritzema, 2001). Further lowering of the water table results in drying of the upper peat layers, impeding peat mineralization and reducing CO_2 emission (Renger et al., 2002).

2.7.5. The effect of agricultural management on peatland CH_4 flux

Agricultural drainage generally decreases CH_4 emissions from peatlands and even turns some of them from CH_4 sources to CH_4 sinks (Knox et al., 2015; Maljanen et al., 2010; Schrier-Uijl et al., 2014), mainly through decreasing the CH_4 production in the anaerobic zone but increasing CH_4 consumption in the aerobic zone. A review of several studies suggested that the average CH_4 emission in agricultural managed peatlands was 0.24 ± 0.48 g C m^{-2} yr^{-1} for Nordic countries (Maljanen et al., 2010). Another meta-analysis showed that drained peatland produces less CH_4 emission by around 8 mg CH_4 m^{-2} d^{-1} when compared with intact peatlands (Bussell et al., 2010). In addition, a few studies which directly compared the CH_4 emission between agriculturally managed peatlands and undisturbed or restored ones indicated that agricultural drainage significantly reduced CH_4 emission of peatlands in USA (Knox et al., 2015), the Netherland (Schrier-Uijl et al., 2014), Germany (Beetz et al., 2013) and Finland (Nykanen et al., 1995). Agricultural management can also regulate peatland CH_4 flux via nitrogen amendment. Ammonium ions were found to be able to inhibit CH_4 oxidation both by competitive inhibition of the enzyme methane mono-oxygenase and through decreasing pH (Hütsch, 1998). In addition to ammonium, nitrate or nitrite ions reduce the activity of CH_4 oxidising bacteria (Reay and

Nedwell, 2004).

2.7.6. The effect of agricultural management on peatland N_2O flux

The emissions of N_2O from undisturbed peatlands are usually negligible, but lowering the water table enhances mineralization of the OM thus triggering the N_2O emissions (Augustin et al., 1998; Martikainen et al., 1993; Merbach et al., 1996). The increase of nitrification and the creation of low molecular humic substances after drainage may be the reason for enhanced N_2O emissions (Lång et al., 1993). Nitrate produced in the upper layers of drained soils as a result of mineralization of humic substance moves to the deeper layers of the profile and is reduced to N_2O (Lång et al., 1993). Drained peatlands are remarkable sources of N_2O with annual flux varying between 2 and 56 kg N_2O ha^{-1}. Great uncertainties exist in the flux estimates due to significantly spatial and temporal variability in N_2O flux in the agriculturally managed peatlands. Van den Pol-van Dasselaar et al. (1998) found the coefficients of spatial variation in N_2O flux ranged from 170 to 500%, resulting from the difference in properties of the peat, fertilization rate, temperature and soil moisture and WT. Langeveld et al. (1997) reported high N_2O emission of 14 to 61 kg ha^{-1} yr^{-1} 1-3 weeks after fertilizer application, but very low fluxes in summer during the dry period at pastures in the Netherland. High N_2O emissions were found when high soil moisture occurred in drained fens in Germany and Finland (Augustin et al., 1998; Nykänen et al., 1995). Maljanen et al. (2002) found soil temperature was the main factor regulating the emissions of N_2O and thus the highest emissions in agricultural soils occur in the afternoon. Since the mineralization of peat OM itself produces substrate for denitrification, direct fertilization effects are not always seen (Augustin et al., 1998; Regina et al., 2004). With very high application rates (480 kg N ha^{-1}) the fertilization effect can be seen, however the effect was not obvious under low application rate (Augustin et al., 1998). As in mineral agricultural soils,

also in peat soils a remarkable part of the annual flux can occur outside of the vegetation period when there is no plant uptake of nitrogen. The weather conditions, e.g. the frequency of freeze-thaw cycles, thickness of snow cover and possible ice layers, partly determine the magnitude of the winter fluxes (Maljanen et al., 2003a; Regina et al., 2004).

2.8. Conclusion

Northern peatlands have stored a large quantity of C, playing an important role in the global C cycle and regional climate regulation. Many researchers have studied the fluxes of C and greenhouse gases in peatlands. However, few studies have considered the fluxes of all GHGs (CO_2, CH_4 and N_2O) at the same time when budgeting the C and GHGs fluxes of peatlands. Usually, CH_4 is mostly ignored when estimating the C balance and N_2O is usually neglected when quantifying the GHGs flux in peatlands, contributing to the bias in estimating and predicting either the C or GHGs balance and their feedbacks to changing climate. In addition, many studies about how the related factors affect peatland C cycling are based on controlled laboratory experiments, making it difficult to estimate the real responses to the regulators at the field scale. Thirdly, the measurements of C cycling processes based on chamber techniques seldom allow for a clear understanding of the processes and controls that underlie the observed net processes at the ecosystem scale. Moreover, large areas of peatlands, with an estimated area of 300×10^3 km^2 have been drained for agricultural development in the past decades (Joosten and Clarke, 2002). Although great attention has been paid on the effect of land management on peatland C and GHGs flux, hitherto, limited studies have compared directly the C and GHGs cycling processes of undisturbed peatland and agriculturally managed peatland directly at the ecosystem scale. The lack of comprehensive studies makes it impossible to draw clear conclusions about the effects of peatland management on the overall C balance (Haddaway et al.,

2014).Canada contains 1.2 million km^2 of peatlands and agricultural use is the major management practice for peatlands (Joosten and Clarke, 2002). The role of climate, hydrologic, and ecologic controls in GHGs cycling of peatlands in Canada have been studied for a long time at field and micro-topographic scales. Such studies provide important insights for understanding the factors on peatland C cycling processes and predicting their feedback to climate change and human activities. However, most of this work have been focused on the peatlands in southeastern Canada at the Mer Bleue, near Ottawa, ON and sites near Riviere du Loup, QC and Shippagan, NB (Bubier et al., 1993; Frolking et al., 1998; Joiner et al., 1999; Lafleur et al., 1997, 2005a,b; Moore et al., 2002; Rayment and Jarvis, 2000; Roulet et al., 2007; Waddington and Roulet, 2000), few studies are carried out at peatlands in other regions of Canada, such as Newfoundland. Studying the C and GHGs cycle processes and the corresponding controls as well as their response to land management of peatlands in Newfoundland is intended to improve the understanding of peatland C and GHGs cycle processes in Canada, thus assisting in making more accurate estimation and prediction of the C fate as well as their climate effect in peatlands at the national scale.

Chapter 3 - Temporal shifts in controls over methane emissions from a boreal bog in western Newfoundland, Canada

3.1. Abstract

We measured year-round landscape-scale methane (CH_4) flux in a boreal bog from May 2014 to April 2016 using the eddy covariance technique. The objectives of the study were to investigate the controls on CH_4 flux over different time scales from diel to seasonal periods and to quantify the annual CH_4 flux budget. We observed strong diel pattern of CH_4 flux only during the growing season in both 2014 and 2015, whereby CH_4 emission increased from nighttime to daytime and peaked near mid-daytime. Diel pattern of CH_4 flux was mainly related to the variation in photosynthetically photon flux density in 2014 and surface soil temperature at 1 cm in 2015. Strong seasonal variability in the daily average CH_4 fluxes was observed in both 2014 and 2015, ranging from near zero before May to a peak of above 20 nmol m^{-2} s^{-1} in the middle-late August in 2014 and in the early-middle September in 2015. Soil temperature at 50 cm was the dominant control over the seasonal variation in the daily average growing season CH_4 flux in both years, though water table depth also exerted some influence in 2014. During the non-growing season, the daily variation in CH_4 fluxes was mostly related to friction velocity in both years. In addition, daily average CH_4 flux increased with enhanced net ecosystem exchange of CO_2 (NEE) when NEE was negative (i.e., days when CO_2 was being taken up), but there was no correlation between them when NEE was positive (days with net CO_2 emissions) during the growing season. We found that this boreal bog acted as a small CH_4 source of 3.58 $\pm$ 0.69 g CH_4 m^{-2} from May 2014 to April 2015 and 3.07 g $\pm$ 0.67 CH_4 m^{-2} from May 2015 to April 2016. These values were

at the lower end of the range of CH_4 emission rates reported for boreal peatlands. Non-growing season CH_4 emissions accounted for 47% (the first study year) and 33% (the second study year) of the annual emissions, highlighting the importance of non-growing season CH_4 emissions in estimating the annual CH_4 budget and the feedback to climate.

3.2. Introduction

Methane (CH_4) is an important greenhouse gas, with the global warming potential about 25 times that of carbon dioxide (CO_2) on a 100-year time horizon (IPCC, 2014). The atmospheric CH_4 concentration has increased by 148%, from 715 ppb in pre-industrial times to 1774 ppb in 2005, and this trend is expected to continue in the future, thus it can exert a great impact on the future climate system (IPCC, 2014). Indeed, the persistent increase in concentrations of atmospheric CH_4 has accounted for 20% of the total increase in radiative forcing over the past century (IPCC, 2014) and motivates efforts to understand the driving forces of different global CH_4 sinks and sources.

Northern peatlands are presently a sink of CO_2 but a source of CH_4 (Mikaloff Fletcher et al., 2004; Roulet, 2000). The ratio of CO_2 absorption to CH_4 emission largely determines the influence of peatlands on climate (Frolking et al., 2006). The effect of CH_4 emission variation was suggested to dominate the radiative forcing impact of peatlands on climate in the first few decades after their formation, after which the impact of the change in CO_2 sequestration slowly gained significance (Frolking et al., 2006). Currently, CH_4 emissions from peatlands contributes ~3-5% of the total global CH_4 emission (Mikaloff Fletcher et al., 2004; Prather et al., 2001), which offsets the "climate cooling" effect due to CO_2 uptake by peatlands (Frolking and Roulet, 2007). Therefore, CH_4 flux should be taken into consideration when prescribing the carbon (C)

and greenhouse gas (GHGs) budgets of peatlands.

Although great efforts have been made in quantifying CH_4 emission rates and their contribution to the greenhouse gases balance of peatlands (Pypker et al., 2013) and in addressing the controls on the spatial and temporal variability in CH_4 flux (Abdalla et al., 2016; Frenzel and Karofeld, 2000; Granberg et al., 1997; Günther et al., 2014; Lai et al., 2014; Liblik et al., 1997; Moore and Knowles, 1989; Treat et al., 2007; Turetsky et al., 2014). Previous research has suggested that the CH_4 emission rates ranged over three orders of magnitude among northern peatland ecosystems (Koebsch et al., 2015; Turetsky et al., 2014; Vanselow-Algan et al., 2015; Zhuang et al., 2006) and the controls on CH_4 flux vary among different peatland types (Abdalla et al., 2016; Turetsky et al., 2014), among microtopographies within the same peatland (Bubier et al., 1993; Nilsson et al., 2001; Skov, 2014) and at different timescales (Günther et al., 2014; Koebsch et al., 2015). The wide fluctuations in previously measured CH_4 emission rates and changing driving forces pose challenges in accurately estimating the CH_4 budget of peatlands at a global scale and for model development and testing (Bridgham et al., 2013). Indeed, estimated regional CH_4 emission rates of northern peatlands range from 32 to 112 Tg CH_4 yr^{-1} (1 Tg = 10^{12} g) (Bergamaschi et al., 2007, 2009; Petrescu et al., 2010; Zhuang et al., 2004). Moreover, existing CH_4 biogeochemistry models have inadequately incorporated the controls over CH_4 production, transport and consumption (Bridgham et al., 2013). Therefore, more information on the controls on CH_4 fluxes dynamics of peatlands are needed in order to better predict and project their contribution to global atmospheric CH_4 budget and their feedback to future changing climate.

Although not a universal finding, many previous studies have found strong diel (Gažovič et al.,

2010; Long et al., 2010; Suyker et al., 1996), seasonal (Dise, 1993; Long et al., 2010; Riera et al., 1999; Shurpali and Verma, 1998; Shurpali et al., 1993) and inter-annual (Shannon and White, 1994; Treat et al., 2007) variation in peatland CH_4 flux. However, the controls over CH_4 flux are dynamic and change at different timescales (Günther et al., 2014; Koebsch et al., 2015). Soil temperature and water table depth are generally found to exert the greatest impact on the CH_4 emission dynamics at varying timescales (Frenzel and Karofeld, 2000; Granberg et al., 1997; Günther et al., 2014; Koebsch et al., 2015; Long et al., 2010; Mikkelä et al., 1995; Moore and Knowles, 1989; Pypker et al., 2013; Treat et al., 2007; Turetsky et al., 2008). In addition, peatland plants are an essential regulator of the temporal variability in CH_4 fluxes through changes in the substrate availability for CH_4 production (Koebsch et al., 2013; Pypker et al., 2013) and through plant-driven gas transport (Chen et al., 2010; Kowalska et al., 2013). Other factors such as vapor pressure deficit (VPD) (Long et al., 2010), friction velocity (Gažovič et al., 2010), solar radiation (Long et al., 2010; Mikkelä et al., 1995), latent heat flux (Long et al., 2010), and ecosystem conductance of water vapor (Long et al., 2010) have also been observed to be correlated with the diel to seasonal variations in CH_4 flux.

In this study, we report two years of landscape-scale CH_4 fluxes measured by eddy covariance at an undisturbed boreal bog in an attempt to investigate the diel, seasonal and inter-annual variation of the CH_4 flux, to identify the controls on CH_4 flux at different timescales and quantify the annual CH_4 budget. We hypothesized that 1) there would be a strong diel, seasonal and inter-annual variations in the CH_4 flux, 2) the controls over CH_4 fluxes would vary at these different timescales, and 3) this bog would act as an annual CH_4 source as suggested by previous chamber measurements at the bog (Luan and Wu, 2015).

3.3. Materials and Methods

3.3.1. Study site

Our site, an undisturbed boreal bog, is located in the Robinsons pasture western Newfoundland, Canada (48.260 N, 58.663 W). According to the data from the nearest weather station in Stephenville (48.541 N, 58.55 W) (http://climate.weather.gc.ca/climate_normals/results_1981_2010_e.html?stnID=6740&autofwd=1), the average annual temperature of 1981-2010 is estimated at approximately 4.5 ^{o}C and the annual rainfall is ~955 mm. Details of site characteristics can be found in Luan and Wu (2014), and are only described briefly here. The bog, about 200 ha, consists of different landforms mainly including peatland pools, hollows and hummocks. The peatland pools varied significantly in size, ranging from 10 to 200 m^2, and were permanently inundated with the standing water depth of 40-60 cm. They cover only about 10% of the bog. The hollow and hummock landforms have a substrate mostly covered with bog moss species *(Sphagnum warnstorfii and Sphagnum capillifolium)* and partly with gray reindeer lichens *(Cladina spp.)*. Sedges *(Trichophorum cespitosum)* and herbs *(Muhlenbergia rigens, Rubus chamaemorus and Lycopodiopsida)* dominate the wetter hollow but ericaceous shrubs *(Gaylussacia spp., Empetrum nigrum, Kalmia angustifolia, Rhododendron groenlandicum, Cornus canadensis and Andromeda polifolia*) mainly occur on the drier hummock. Hollows and hummocks each account for about 45% of the bog area. The dry aboveground biomass, measured in 2013, was 197 ±87 g m^{-2} in hummocks, similar to that of 191 ±41 g m^{-2} in hollows. However, the dry root biomass was 745 ±200 g m^{-2} in hummocks, significantly higher than that of 421 ±141 g m^{-2} in hollows ($P<0.05$) (Luan and Wu, 2015).

3.3.2. CH_4 flux and meteorological measurements

CH_4 flux measurements were made with an eddy covariance (EC) system (Fig.3.1). Wind vectors (u, v, w) and sonic temperature were measured with a three-dimensional (3-D) sonic anemometer (Gill WindMaster Pro, Gill Instruments) mounted at 3.44 m height above the mean surface of the bog. An open path infrared gas analyzer (LI-7700, LI-COR Inc., Nebraska, USA) mounted at 3.34 m height, with the separation from sonic anemometer of 17cm northward, -1cm eastward and 10cm vertically to measure CH_4 molar density. A fast response infra-red gas analyzer (IRGA: LI-7200 Enclosed CO_2/H_2O Analyzer, LI-COR Inc., Nebraska, USA) was used to measure variations in CO_2 and H_2O molar densities. The LI-7200 analyzer was mounted at the height of 3.21 m with the separation between the sonic and IRGA being 3.5 cm northward, 3.5 cm eastward and 23 cm vertically. Air was pulled by a diaphragm pump through a 1 m long sample tube to the IRGA at a rate of 16.1 L min^{-1}. Instantaneous CO_2 and H_2O concentrations were measured inside the sampling cell, along with instantaneous air temperature and air pressure. The enclosed LI-7200 analyzer outputs not only instantaneous gas density for traditional flux calculations (Webb et al., 1980), but also instantaneous mixing ratio of CO_2 and H_2O, which use instantaneous water, temperature and pressure measurements inside the cell to correct for dilution, temperature and pressure. Two thermocouples were used to measure the instantaneous temperatures of air just before entering and exiting the sampling volume. A differential pressure sensor with a high speed and precision, together with a low speed, high precision absolute pressure sensor were used to measure instantaneous pressure in the middle of the cell. Data output from the EC system instruments were recorded at 10 Hz with a data logger (LI-7550, LI-COR Inc., Nebraska, USA) and stored on a removable USB device.

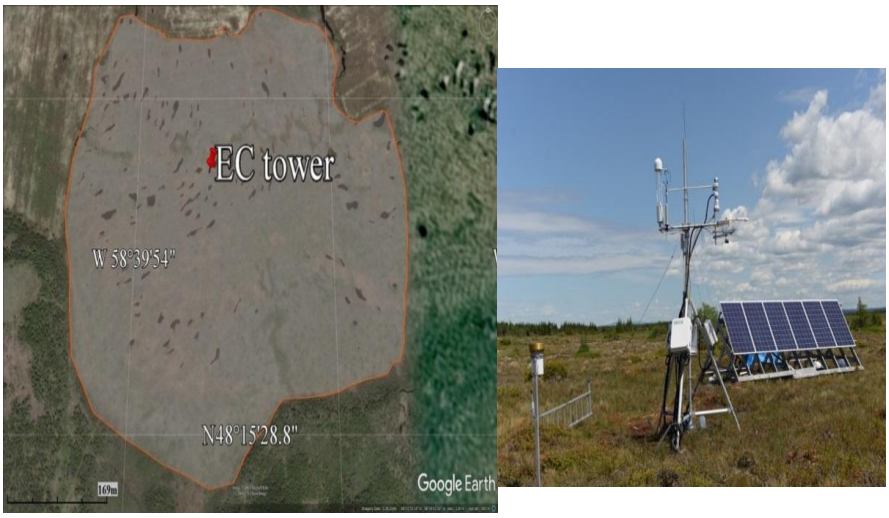


Fig.3.1 Satellite image of the natural boreal bog at the Robinsons pasture, western Newfoundland, Canada (48.260 N, 58.663 W). The image was obtained from Google Earth with imagery collected on May 28, 2006. The outline of the site was indicated by the red solid line and the red pin represents the location of eddy covariance (EC) tower (left); (right) a photo of the setup of EC measurement system.

Environmental variables were recorded by a series of meteorological instruments mounted on the EC system tower. Photosynthetically active photon flux density (PPFD) was measured by quantum sensor (LI-190SL-50, LI-COR Inc., Nebraska, USA). Air temperature (T_a) and relative humidity (RH) was measured with a temperature / relative humidity probe (HMP155, Vaisala, Vantaa, Finland). A 4-component net radiometer (CNR4, Kipp & Zonen, Delft, The Netherlands) was amounted at a height of ~3.5 m to monitor the incoming and reflected short-wave and long-wave radiation. A tipping-bucket rain gauge (TR-525USW, Texas Electronics, Texas, USA) was mounted on the ground to measure rainfall. Soil temperature was measured at depths of 1 cm, 5 cm, 10cm, 30 cm and 50 cm below *Sphagnum* moss surface by thermistors (LI7900-180, Li-Cor Inc., Nebraska, USA) and soil moisture was measured as volumetric water content at 5 cm, 10

cm, 30 cm and 50 cm with a water content probe (Delta-TML2x, Delta-T Devices, U.K.). Water table (WT), defined by using the surface of mosses as the zero datum, was monitored by a stainless steel pressure transducer sensor with SDI-12/RS232 connection (CS451, Campbell Scientific, Utah, USA). Rainfall was recorded as 30-min totals, and all other environmental variables were scanned at 5-s intervals and recorded as half-hourly means on a data logger (CR3000-XT, Campbell Scientific, Utah, USA).

3.3.3. Data Processing

Data quality control, and gap filling and partitioning

We used EddyPro 5.2.1 software (LI-COR Inc., Nebraska, USA) to process the 10 Hz raw data from the EC system and output corrected fluxes of CH_4 and CO_2 over a 30-min interval. In this study default settings were applied to adjust for air density fluctuations by Webb-Pearman-Leuning for CH_4 flux (Webb et al., 1980) and converting to mixing ratio for CO_2 flux (Burba et al., 2012), corrections for frequency response (Moncrieff et al., 1997, 2004), double axis rotation for sonic anemometer tilt correction (Wilczak et al., 2001), angle-of-attack correction for wind components (Nakai and Shimoyama, 2012), statistical control tests for fluxes (Vickers and Mahrt, 1997), quality control tests for fluxes (Mauder and Foken, 2011), flux footprint estimation (Kljun et al., 2004), lag minimization using maximum covariance with default lag of 0, and calculation of friction velocity (u*) using both along and cross wind shear. The data quality of the corrected half-hourly fluxes were indicated by the values of diagnostic flags, with 2 representing data of bad quality (Mauder and Foken, 2011) and these data were discarded. The half-hourly flux data with received signal strength values (RSSI) smaller than 20% and those during periods of rainfall were also discarded. The final flux data was corrected by adding the flux storage below the height of the IRGA, computed from half-hourly mean concentration changes at the sensor level.

Fetch in the bog varied from about 270 m to 400 m in different directions. We used this information to filter the fluxes as follows. Data with the 70% footprint larger than 380 m with wind direction of 0-180 ºand larger than 300 m in the sector 180-360 ºwere rejected in order to guarantee the fluxes source are mostly from within the bog boundary. In addition, we did not find any relationship between u* and CH_4 flux, so we discarded CH_4 flux with u* less than 0.1 m s^{-1} according to a previous study (Reichstein et al., 2005). Approximately 58% (2014) and 45% (2015) of the growing season CH_4 flux data and 53% of the non-growing season flux data in both study years were rejected due to bad data quality and instrument failure.

Gaps in the CH_4 flux data were fill using an artificial neural network (ANN), this method is one of a suite of tools being used for gap-filling in flux studies (Moffat et al., 2007; Papale et al., 2006) and has been shown recently to be highly successful for gap-filling CH_4 fluxes (Dengel et al., 2013). The ANN was performed using the Matlab numerical software. Data were divided into clusters of daytime and nighttime according to the PPFD threshold of 20 μmol m^{-2} s^{-1}. 70% of available date in each cluster was used to train the network, an additional 15% for testing the network and finally, 15% for validating the ANN. Before training, all data were normalized as 0-1 as in (Aubinet et al., 2012; Dengel et al., 2013; Moffat et al., 2010). The architecture of each neural network was initialized 10 times with random starting weights, and the initialization resulting in the lowest mean sampling error was used (Järvi et al., 2012). The simplest architecture, whereby additional increases in complexity resulted in less than 5% increase in mean square error, was selected and the prediction was saved. Therefore, we set the number of neurons in the fitting network's hidden layer as 10. This procedure was replicated for 20 times

and the median predictions were used to fill missing half hour fluxes. The neural network includes an input layer, a hidden layer and an output layer (Elizondo and Góngora, 2005; Jain et al., 1996) and this two-layer feed-forward network with sigmoid hidden neurons and linear output neurons can fit multi-dimensional mapping problems arbitrarily well. The ANN was trained with the Levenberg-Marquard backpropagation algorithm in MatLab (trainlm), as used in previous studies (Dengel et al., 2013; Riedmiller.,1994). We chose input variables including air temperature, soil temperature at 1 cm and 50 cm, solar radiation, vapor pressure deficit (VPD), u* and WT according to a previous study (Dengel et al., 2013). However, during some periods in wintertime, VPD and u* data were also missing, so we only used the remaining variables mentioned above to fill the data gaps at these times. Gap-filled flux data were only used for CH_4 budgeting in this analysis.

We partitioned CH_4 fluxes into four periods as Song et al. (2015): the growing season and three non-growing seasons including soil thawing, soil freezing and winter. We assumed growing season began and ended after the first seven consecutive days with daily air temperature above 5°C and below 5°C, respectively. We further subdivided the growing season into three periods of early growing season (May and June), peak growing season (July and August) and late growing season (September, October and November). Soil freezing, ranging from the end of the growing season to the first two consecutive days with average daily soil temperature below $0^{o}C$ at 5 cm depth. Winter started at the end of the soil freezing period and ended when snow melted out (after seven consecutive days with average air temperature above $0\ ^{o}C$). The soil thawing period was between winter and the growing season. The purpose of partitioning CH_4 into different periods was to estimate the contribution of cumulative CH_4 flux in each period to the annual flux

budget as well as to examine the variations in the controlling factors of CH_4 flux in different growing season periods and the non-growing season. We did not find any significant CH_4 burst during the soil freezing and thawing periods and the data availability of these two periods were limited. Therefore, we lumped all non-growing season period together to identify the environmental controls on the CH_4 flux.

3.3.4. Flux uncertainty estimation

Although there are many sources of uncertainty in flux estimation measured by eddy covariance, here we focused on flux random uncertainty due to sampling errors. The other uncertainty sources [e.g. errors due to the buoyancy effects of heat and water vapor, errors due to limited response time of the sensors, errors due to separation of the sensors, errors due to random noise in the system, errors due to inadequate or excessive height of the sampler over the surface, errors due to inadequate fetch, errors due to flow distortion caused by aerosol particles, and etc.] can be avoided due to either carefully and properly field experiment design (Businger, 1986) or flux data correction as mentioned above, sampling error will remain as one of the largest sources of uncertainty. Flux random uncertainty (σ_1) in EddyPro is calculated following Finklestien and Sims (2001). This method requires the preliminary estimation of the Integral Turbulence Time-Scales, which can be defined as the integral of the cross-correlation function between vertical wind component and any scalar of interest (e.g. temperature, gas concentration, etc.), details can be found in (Finkelstein and Sims, 2001). We also estimated the flux uncertainty due to gap-filling (σ_2) based on the following procedures. Firstly, we developed, trained and validated ANN model using the available measured data in each study period (i.e., growing season, soil freezing period, soil thawing period and wintertime). Secondly, we ran the ANN model and produced a continuous series of data for the whole two-year study period. Finally, we compared the

available measured data and their counterpart predicted CH_4 flux values from ANN model in each study period (Moffat et al., 2007). $\sigma_2 = 1 / N \sum(P_i - O_i)$. N is the number of measured and predicted CH_4 flux pairs in a certain period and P_i and O_i are the individual predicted CH_4 flux data and the observed value, respectively. The total uncertainty was calculated following the equation: $\sigma = [\sigma_1^2 + \sigma_2^2]^{1/2}$.

3.3.5. Statistical analysis

We studied the diel pattern of CH_4 flux and related environmental variables for each period of both study years. Stepwise multi-variable regression analysis was applied to further determine the contribution of each variable including T_a, peat temperature at 1 cm (T_1), u*, VPD and PPFD to the diel variation in CH_4 flux in the whole growing season of 2014 and 2015. On a seasonal basis, we conducted stepwise multi-variable regression analyses to determine the relative importance of these biotic and abiotic variables including VPD, u*, air temperature, soil temperature at different depths, PPFD, WT and net ecosystem exchange of CO_2 (NEE) in controlling the temporal variation in daily CH_4 flux in different study periods. These analyses were conducted using the statistical program SAS v9.1. All statistical analyses were performed only on the measured data in days with at least 70% data available. All data were normalized between 0 and 1 so that data of different units or orders of magnitude can be compared and weighted. Data were normalized as follows: (data - mindata) / (maxdata - mindata); where mindata and maxdata represent the minimum and maximum value of each variable, respectively.

3.4. Results

3.4.1. Seasonal variation in environmental variables

Air temperature during the study tended to be close to long-term normal (within ± 1 standard deviation of long-term averages) with the exceptions of higher than normal temperature in July 2014 and August 2015 and lower than the long-term averages in March and April of 2015 (Fig.3.2a). Daily air temperatures ranged from -16.6 ℃ to 23.6 ℃ in 2014-15 peaking in the middle of July in 2014 and from -11.2 ℃ to 21.6 ℃ in 2015-16 with the maximum values in middle-late August and the lowest values in the end of February of both years (Fig. 3.3a). Overall, annual air temperature averaged 4.4 ℃ in 2014-15 and 4.7 ℃ in 2015-16 (Table 3.1). The seasonal pattern of soil temperature was similar to that of air temperature, except for a leveling-off of minimum temperatures between January and April, while the peatland was snow covered (Fig. 3.3b). The coldest subsurface soil temperature (at 50 cm, T_{50}) occurred at the end of April when snow melted out and the warmest occurred near early-middle August in 2014 and in early September in 2015, almost 2 weeks later than of the maximum air temperature (Fig. 3.3b).

The average daily WT in hollow ranged from -0.33 to -0.06 m in 2014-15 and from -0.36 to -0.07 m in 2015-16, with several episodes of rapid rises and drops in both years following summer rainfall events (Fig.3.3d, e). The drier than normal conditions of September 2015 resulted in decline of ~8 cm in the monthly average WT when compared with that of 2014 (Fig.3.2b, 3.3d, Table S3.1). The yearly average WT was -0.18 m in 2014-15 and -0.21 m in 2015-16. Soil moisture at 30 cm followed a similar pattern as WT and ranged from ~0.76 to 0.88 m^3 m^{-3} over the study period (Fig.3.3c, d). Yearly cumulative rainfall was 942 mm in 2014-15 and 890 mm in 2015-16 (Table 3.1), comparable to the long-term average (955 ± 133 mm).

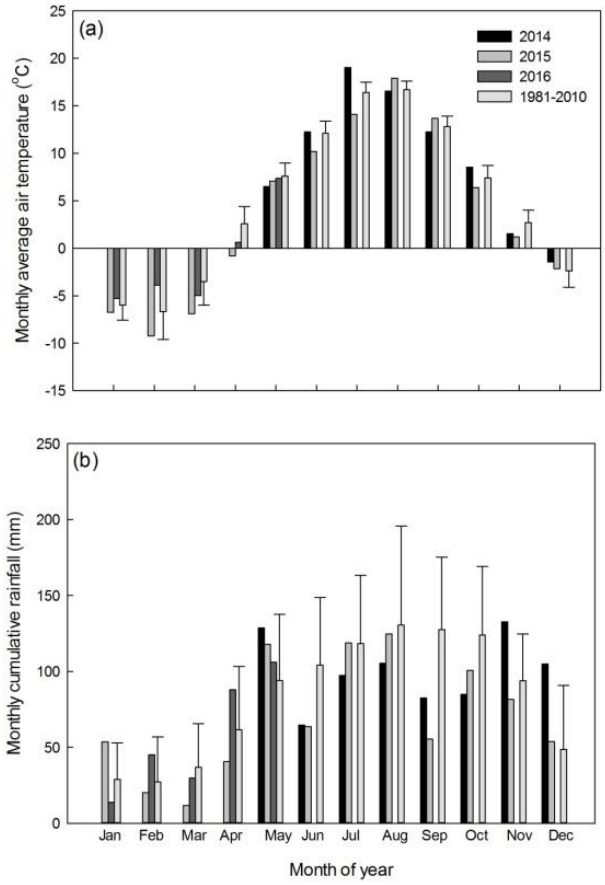


Fig.3.2. Comparison of monthly average temperature and cumulative monthly rainfall measured at the pasture during measurement periods from April, 2014 to May, 2016 with the long-term (30 years average ± SD) measurement of the nearby climate station at Stephenville, Newfoundland and Labrador during 1981-2010.

Table 3.1 Average daily air temperature, soil temperature at depth of 1cm, 10 cm and 50 cm, photosynthetic photon flux density (PPFD), cumulative rainfall, and water table for four different periods in the two study years

Period		Date	Air temperature	Soil temperature (℃)			PPFD	Rainfall	WT
			(℃)	1cm	10 cm	50 cm	(mol m^{-2} d^{-1})	(mm)	(m)
Growing season	Early	2014.5.15-6.30	11.4	12.4	9.4	5.9	38.2	81	-0.21
	Middle	2014.7.1-8.31	18.0	18.5	16.9	12.7	35.5	203	-0.23
	Late	2014.9.1-11.8	9.8	10.4	11.1	11.4	18.6	229	-0.19
	Overall	2014.5.15-11.8	13.1	13.7	12.7	10.4	29.7	513	-0.21
	Early	2015.5.16-6.30	10.2	10.9	9.3	6.2	33.6	161	-0.18
	Middle	2015.7.1-8.31	16.1	17.0	15.3	11.5	32.1	244	-0.25
	Late	2015.9.1-11.15	8.5	9.3	10.4	11.0	14.1	181	-0.24
	Overall	2015.5.16-11.15	11.5	12.3	11.8	10.0	25.1	585	-0.23
Soil freezing		2014.11.9-2015.1.9	-1.9	1.0	2.7	5.4	5.0	191	-0.15
		2015.11.16-12.30	-1.2	0.7	2.4	4.9	4.9	92	-0.21
Winter		2015.1.10-2015.4.30	-5.6	-0.4	0.1	2.1	19.7	126	-0.17
		2015.12.31-2016.4.30	-3.4	-0.1	0.0	1.9	15.5	174	-0.22
Soil thawing		2014.5.1-5.14	2.6	3.7	0.1	1.2	36.4	112	-0.14
		2015.5.1-5.15	3.6	3.5	0.1	1.7	40.0	38	-0.16
Annual		2014.5-2015.4	4.4	6.9	6.7	6.7	22.7	942	-0.18
		2015.5-2016.4	4.7	6.4	6.2	6.3	20.0	890	-0.22

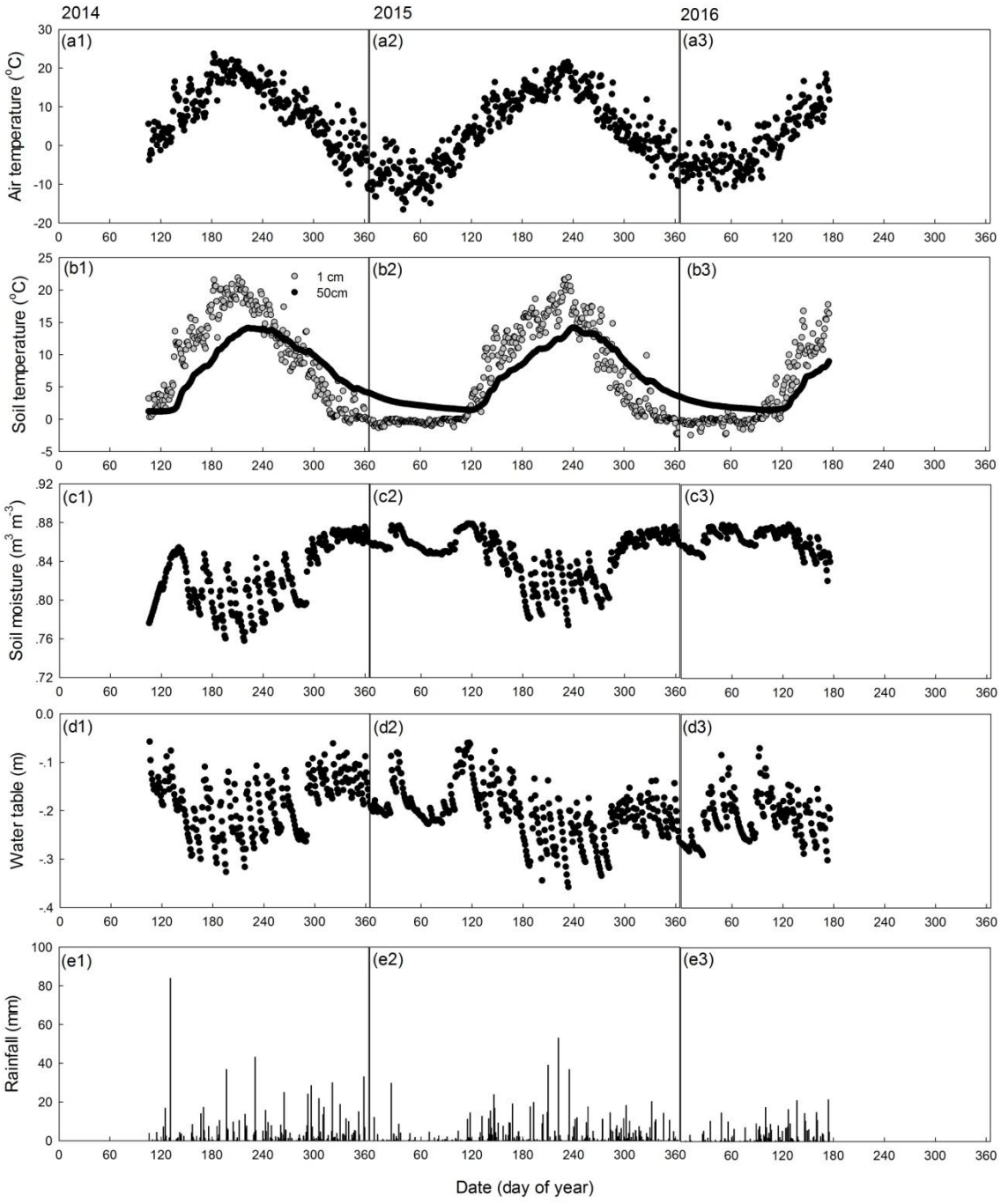


Fig. 3.3 The daily average air temperature (a1-a3), soil temperature at 1 cm and 50 cm (b1-b3), volumetric soil water content at depth of 30 cm (c1-c3), water table (d1-d3) and cumulative rainfall (e1-e3) during the measurement period.

3.4.2. Diel and seasonal variation in CH_4 flux

During the course of the day CH_4 flux was dominated by emissions in both years. However, the diel pattern of CH_4 flux varied among the different seasonal periods, with a strong diel pattern in the growing season, but muted and/or indistinct patterns in the other periods. During both the 2014 and 2015 growing seasons the CH_4 flux increased from nighttime to daytime and peaked after mid-day. In 2014 the peak was sharp occurring around 12:00-14:00, however, in 2015 the daily peak was broader occurring between 13:00 and 18:00 (Figs. 3.4a, 3.5a). The diel pattern for most of the environmental variables showed a similar diel cycle (Figs. 3.4, 3.5), making it difficult to distinguish the real effect of each variable on the diel variation in CH_4 flux. The results of stepwise multi-variable regression analysis suggested that PPFD played a dominant role in regulating the diel variation in growing season CH_4 flux in 2014, explaining 78% of the CH_4 flux variation, and T_1 and T_a also exerted some influence in 2014, together accounting for 8% of the diel variation in CH_4 flux. However, T_1 was the dominant role in the diel variation in CH_4 flux, explaining 64% of the CH_4 flux variation in 2015 (Table 3.2).

Daily average CH_4 flux showed strong seasonal variability in both 2014 and 2015 (Fig.3.6). The CH_4 emission rates ranged from near zero in early May to a peak of above 20 nmol m^{-2} s^{-1} occurring in the mid-late August in 2014 and in early-mid September in 2015 and then decreased to near zero by the end of the growing season. The growing season CH_4 flux peak occurred about ten days later than the maximum subsurface soil temperature at 50 cm (Figs. 3.3b & 3.6). During the non-growing season periods, CH_4 fluxes were quite noisy, ranging from approximately -10 to 20 nmol m^{-2} s^{-1}. There were no observed pulses in daily average CH_4 flux during soil freezing or thawing periods.

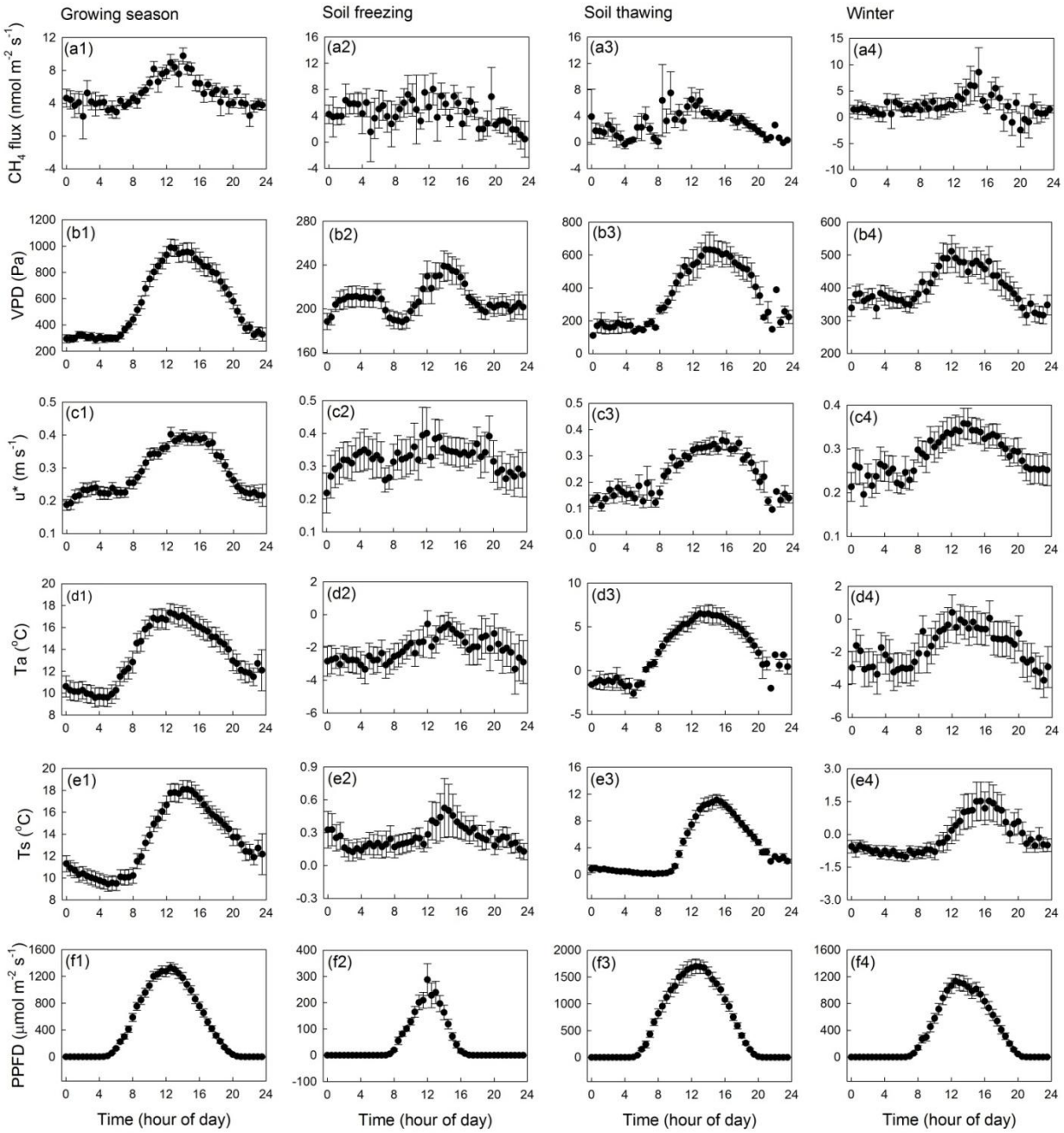


Fig.3.4 The diel variation in methane net flux and environmental variables in different periods in 2014. CH_4 flux (a1-a4), vapor pressure deficit (VPD) (b1-b4), friction velocity (u*) (c1-c4), air temperature (d1-d4), soil temperature at 1cm (e1-e4) and photosynthetically photon flux density (PPFD) (e1-e4). Error bar indicates standard error of the average.

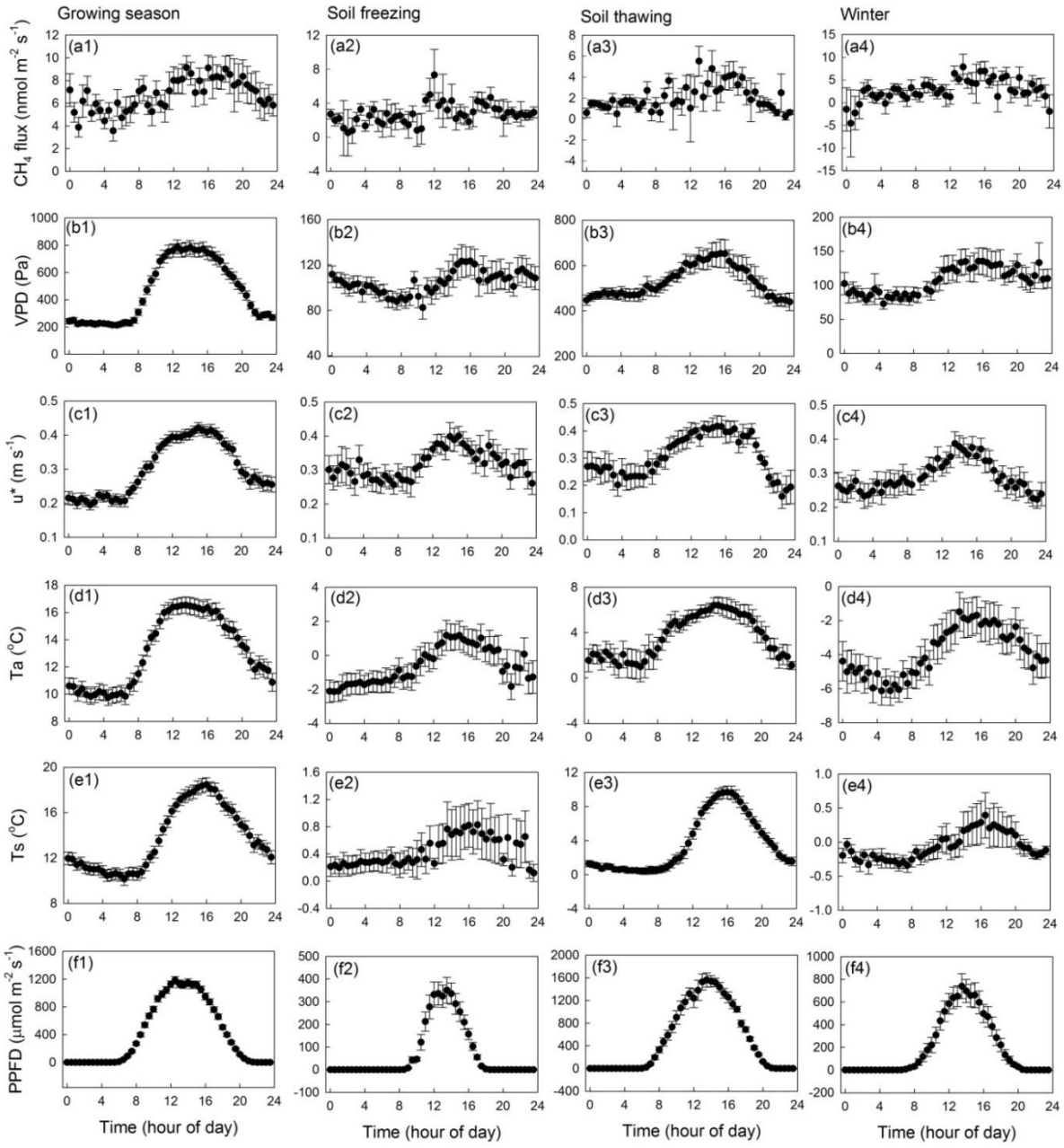


Fig.3.5 The diel variation in methane net flux and environmental variables in different periods in 2015: CH_4 flux (a1-a4); vapor pressure deficit (VPD) (b1-b4), friction velocity (u*) (c1-c4), air temperature (d1-d4), soil temperature at 1cm (e1-e4) and photosynthetically photon flux density (PPFD) (e1-e4). Error bar indicates standard error of the average).

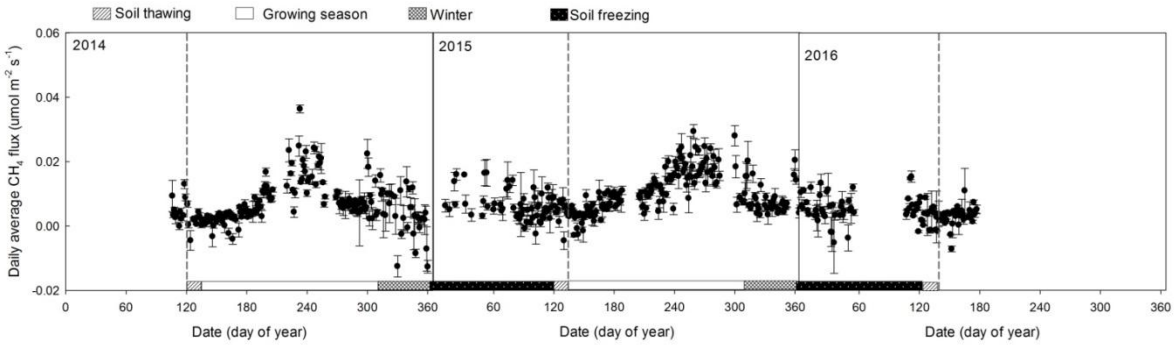


Fig.3.6 The daily average CH_4 flux for different seasonal periods over the two measured years. Whiskers indicate standard error of the average.

Table 3.2 Results of stepwise multi-variate regression relating environmental variables to the diel variation in CH_4 flux during growing season. Environmental variables included in the model were air temperature (T_a), peat temperature at 1 cm (T_1), friction velocity (u*), vapor pressure deficit (VPD) and photosynthetic photon flux density (PPFD). Only significant ($P<0.05$) variables are shown.

Year	Period	Step	Variable entered	Partial R^2	Model R^2	F Value	P
2014	Growing season	1	PPFD	0.78	0.78	162	<0.0001
		2	T_1	0.05	0.83	111	<0.0001
		3	T_a	0.03	0.86	91	0.004
2015	Growing season	1	T_1	0.64	0.64	81	<0.0001

3.4.3. The biotic and abiotic controls over daily CH_4 flux

We found the variables regulating CH_4 flux showed great difference in the different periods of growing season as well as between the two study years (Table 3.3). During the early growing season, CH_4 flux was closely related to T_1 and PPFD in 2014, but to T_{50} and PPFD in 2015. During the peak growing season, CH_4 flux was positively associated with WTand T_{50} in 2014, while WT exerted a negative impact on CH_4 flux in 2015. During the late growing season, CH_4

flux was strongly linked with T_1 in 2014 but with T_{50} in 2015 (Table 3.3). For the whole growing season, T_{50} and WT exerted positive effects on daily average CH_4 flux in 2014, which together explained 52% of the variation in daily CH_4 flux. For the 2015 growing season CH_4 flux was positively related to T_{50} and T_1, but negatively to T_a, which together explained ~69% of the variation in CH_4 flux (Table 3.3). During the non-growing season periods CH_4 flux was poorly related to the environmental variables: u* and T_{50} together explained only 34% of the seasonal CH_4 flux variation in 2014 and u* alone explained 10% of the variation in 2015 (Table 3.3). Moreover, the results from both the stepwise multiple regression and path analysis indicated that T_{50} was the most important factor in regulating CH_4 flux during the growing season and u* gained the importance during the non-growing season in both study years (Table 3.3, Fig.3.7). We found that the relationship between WT and CH_4 varied between 2014 and 2015, especially when WT less than 0.2 m below peat surface (Fig. 3.8a, 3.8b). Since the effect of WT on CH_4 flux can be confounded by soil temperature impact, we classified data into five groups based on T_{50} range (<5 ^{o}C, 5-8 ^{o}C, 8-10 ^{o}C, 10-13 ^{o}C and >13 ^{o}C), an important control in CH_4 flux at this bog, and examined the function of WT on CH_4 flux for each group to exclude the effect of soil temperature. We found that the effect of WT on CH_4 flux depended on soil temperature, whereby the effect was positive when T_{50} was above 13 ^{o}C (Fig.3.8c), but negative when T_{50} less than 5 ^{o}C (Fig.3.8d). No significant relationships between WT and CH_4 flux were found in other temperature ranges, which were not shown here.

Our previous study based on chamber measurements suggested that substrate availability also plays an important role in regulating growing season CH_4 flux (Luan and Wu, 2015). Therefore, we investigated the relationship between NEE (an indicator of substrate availability) and CH_4

flux. Though NEE was not selected in any of the regression models (Table 3.3), we found that the daily average growing season CH_4 flux increased with enhanced daily average CO_2 uptake rate when NEE was negative. However, there was no correlation between CH_4 flux and NEE when the daily average NEE was positive (Fig.3.9).

Table 3.3 The results of stepwise multi-variable regression analysis between CH_4 flux and biotic and abiotic variables, including air temperature (T_a), peat temperature at 1 cm and 50 cm (T_1, T_{50}), friction velocity (u*), vapor pressure deficit (VPD) and photosynthetic photon flux density (PPFD), water table (WT) and net ecosystem exchange of CO_2 (NEE) based on measured daily average data in different study periods and the combined whole growing season.

Period	Sub-period	Year	Model	Variables	Partial R^2	Total R^2	*P*	df
Growing season	Early	2014	$Y = 0.27 + 0.04T_1 + 0.03PPFD$	T_1	0.42	0.42	<0.0001	23
				PPFD	0.18	0.6		
		2015	$Y = 0.27 + 0.04PPFD + 0.05T_{50}$	PPFD	0.17	0.17	0.002	28
				T_{50}	0.14	0.31		
	Peak	2014	$Y = 0.26 + 0.09T_{50} + 0.05WT$	T_{50}	0.63	0.63	<0.0001	25
				WT	0.07	0.07		
		2015	$Y = 0.28 + 0.08T_{50} - 0.11u^* - 0.03WT$	T_{50}	0.3	0.3	<0.0001	39
				u*	0.16	0.46		
				WT	0.06	0.52		
	Late	2014	$Y = 0.30 + 0.05T_1$	T_1	0.47	0.47	0.002	17
		2015	$Y = 0.29 + 0.07T_{50}$	T_{50}	0.36	0.36	<0.0001	28
	All Periods combined	2014	$Y = 0.28 + 0.043T_{50} + 0.025WT$	T_{50}	0.45	0.45	<0.0001	68
				WT	0.07	0.52		
		2015	$Y = 0.32 + 0.06T_{50} - 0.09T_a + 0.05T_1$	T_{50}	0.52	0.52	<0.0001	98
				T_a	0.1	0.62		
				T_1	0.07	0.69		
Non-growing season		2014-15	$Y = 0.29 + 0.08u^* + 0.04T_{50}$	u*	0.26	0.26	<0.0001	55
				T_{50}	0.08	0.34		
		2015-16	$Y = 0.30 + 0.04u^*$	u*	0.1	0.1	0.04	54

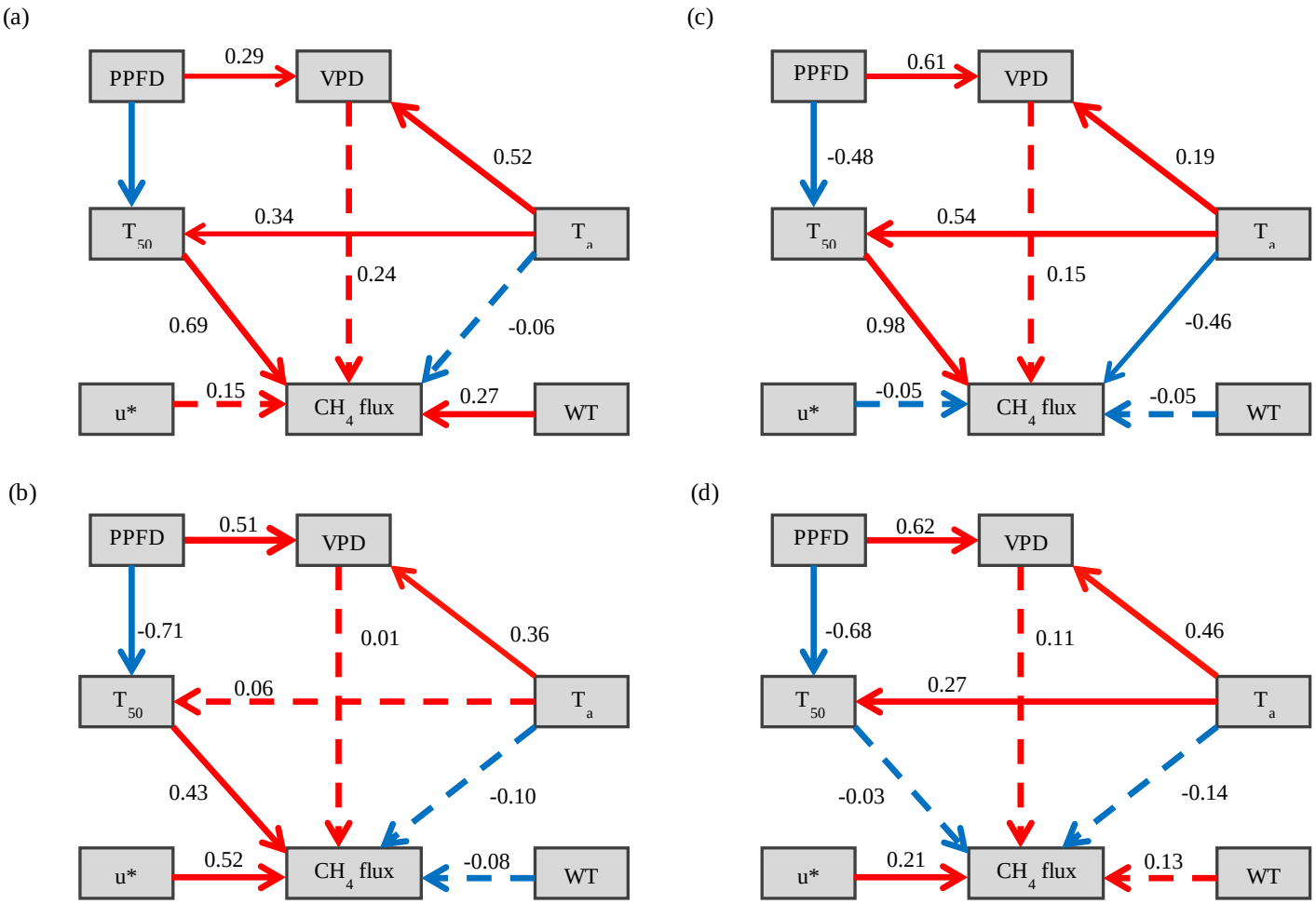


Fig.3.7 Path diagrams illustrating how the environmental variables affect CH_4 flux during growing season in 2014 (a) and 2015 (c), and non-growing season in 2014(b) and 2015 (d). Red arrows indicate a positive effect and blue arrows indicate a negative effect. Numbers besides arrows are the standardized path coefficients. Solid lines indicate relationships with significant impact ($P<0.05$) but dashed lines represent no significant relationship between variables ($P > 0.05$).

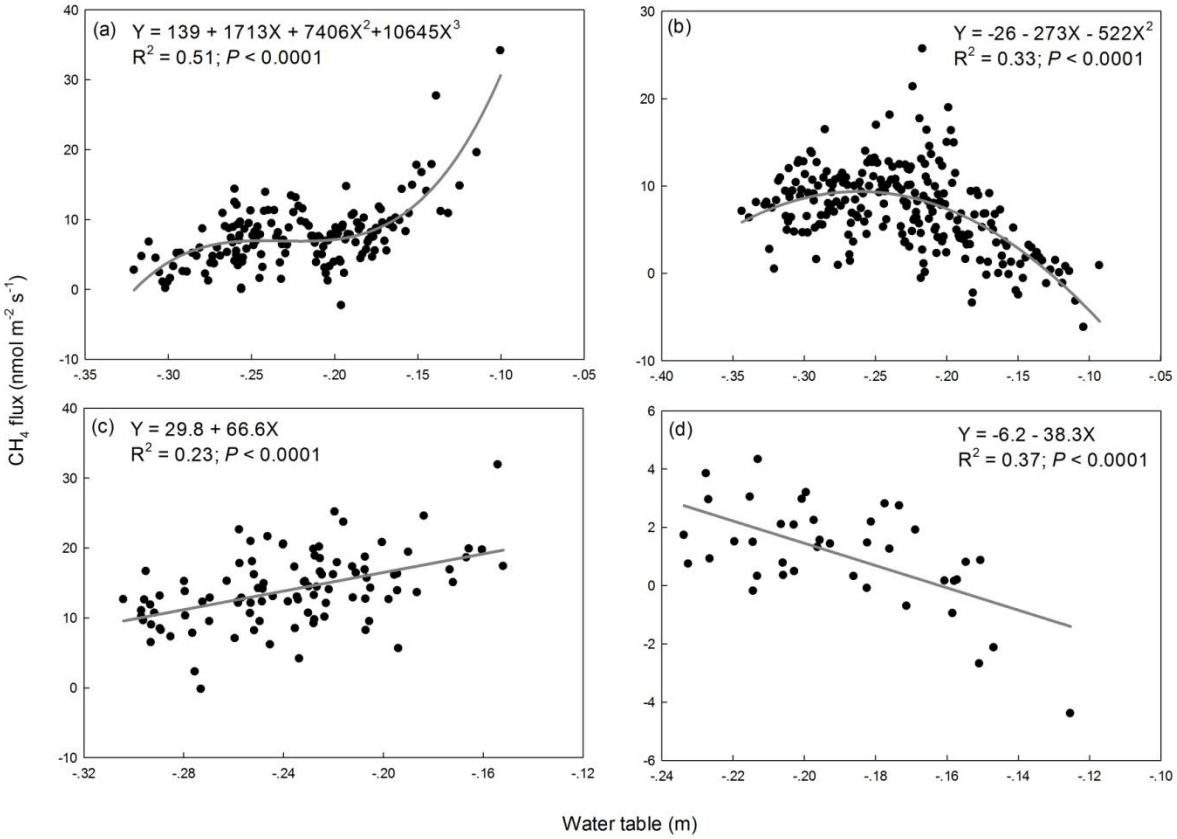


Fig. 3.8 The relationship between the daily average growing season CH_4 flux and water table (a) in 2014, (b) in 2015, (c) when soil temperature at 50 cm (T_{50}) was above 13 ^oC and (d) when T_{50} was below 5 ^oC. Only days with more than 70% data coverage were used. Data from both years were included in panels (c) and (d).

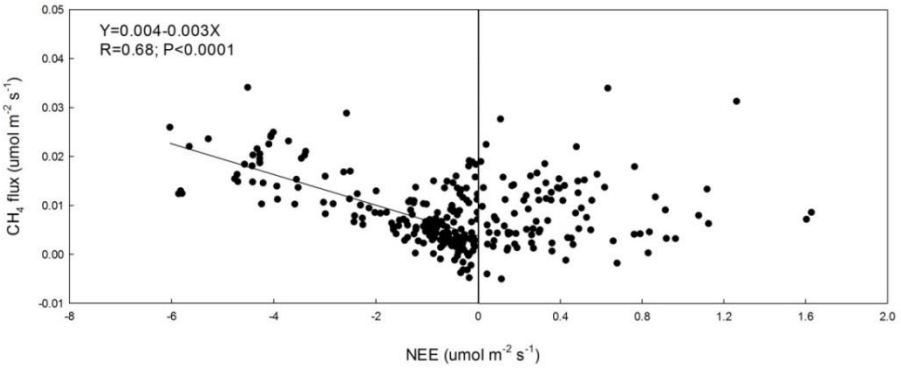


Fig.3.9 Relationship between daily average net ecosystem exchange of CO_2 (NEE) and CH_4 flux during growing seasons of 2014 and 2015. Positive values of NEE indicate emissions of CO_2 and negative values represent CO_2 sequestration from atmosphere.

3.4.4. CH_4 flux budget and uncertainty estimation

The accumulated annual CH_4 flux was ~3.6 ± 0.69 g CH_4 m^{-2} in 2014-15 and 3.07 ± 0.67g CH_4 m^{-2} in 2015-16 according to ANN gap filling (Table 3.4). Most of the flux uncertainty was caused by the random error of about 20% of the cumulative flux and the bias due to gap-filling was small and can be neglected in both years (Table 3.4). Though the growing season CH_4 emissions contributed the greatest amount to the annual total emissions in both years, the non-growing season emissions, which accounted for 47% (2014-15) and 33% (2015-16) of the total, were also important components of the annual budget (Table 3.4).

Table 3.4. Cumulative CH_4 fluxes (g CH_4 m^{-2}) and estimated random uncertainty (RU, g CH_4 m^{-2}) and gap-filling uncertainties (GU, g CH_4 m^{-2}) for the different seasonal periods and their contributions to the annual emissions in two years, May 2014 to April 2015 and May 2015 to April 2016.

	2014-2015					2015-16				
Period	Duration days	CH_4 fluxes	RU	GU	Contribution %	Duration days	CH_4 fluxes	RU	GU	Contribution %
Growing season	178	1.88 ± 0.66	0.66	0.009	53	184	2.04 ± 0.56	0.56	0.0001	67
Soil freezing	62	0.58 ± 0.10	0.1	0.003	16	45	0.25 ± 0.21	0.21	0.02	8
Winter	111	1.06 ± 0.12	0.12	0.006	30	122	0.67 ± 0.27	0.27	0.05	22
Soil thawing	14	0.06 ± 0.13	0.13	0.0003	2	15	0.11 ± 0.15	0.15	0.008	3
Total	365	3.58 ± 0.69	0.69	0.01		366	3.07 ± 0.67	0.67	0.05	

Table 3.5. Comparison of the methane flux between this boreal bog and other undisturbed peatlands

Location				Peatland type	Study Method	CH_4 flux (g CH_4 m^{-2} yr^{-1})			Ref.
Country	Province/City	Latitude °N	Longitude °E			Growing season	Non-growing season	Annual average	
Estonia	Pärnu	58.47	25.21	Temperate bogs	Static chamber			11.3	Salm et al., 2012
Finland	Ruovesi	61.83	24.2	Boreal fen	Eddy covariance	41.3	11.7	16.8	Rinne et al., 2007
Finland	Lapland	69.13	27.27	Arctic mire	Eddy covariance			7.3	Hargreaves et al., 2001
Finland	Ahvensalo	65.85	30.88	Boreal bog	Static chamber		1.3		Alm et al., 1999
Finland	Ahvensalo	62.78	30.93	Boreal fen	Static chamber		4.5		Alm et al., 1999
Finland	Ilomants	62.75	-31.05	Boreal fen	Chamber			34.7	Nykanen et al.,1995
Germany	Swabia	47.81	-11.46	Temperate bog-pine	Eddy covariance			7.1	Hommeltenberg et al.,2014
Poland	Łomża	53.59	22.89	Temperature mire	Eddy covariance				Fortuniak et al., 2017
Russia	Komi Republic	61.93	50.22	Boreal peatland mixture	Static chamber			34.1	Schneider et al., 2016
Siberia	Plotnikovo	57	82	Boreal bog	Static chamber		2.4		Panikov et al., 2000
Sweden	Västerbotten	64.18	19.55	Boreal fen	Static chamber			12, 19	Nilsson et al., 2008
Sweden	Abisko	68.33	19.05	Subarctic palsa mire	Eddy covariance			36	Jackowicz et al., 2010
USA	Michigan	42.45	-84.02	Temperate bog	Static chamber			6.4	Shannon and White, 1994
USA	Minnesota	47.51	-93.49	Temperate poor fen	Eddy covariance			21.7	Olson et al., 2013
USA	Minnesota	47.53	-93.46	Temperate bog	Static chamber			57.3	Dise,1993
USA	Minnesota	47.53	-93.46	Temperate poor fen	Static chamber			87.6	Dise,1993
USA	Minnesota	47.32	-93.47	Temperate bog	Chamber			49.3	Crill, et al., 1988
USA	New Hampshire	43.21	-71.06	Temperate poor fen	Static chamber	152			Treat et al., 2007
USA	Michigan	46.32	-86.05	Sub-boreal	Eddy covariance	17.3			Pypker et al., 2013
Canada	Quebec	53.68	-78.17	Boreal bog	Eddy covariance	28			Nadeau et al., 2013
Canada	Ontario	45.68	-75.8	Temperate bog	Chamber&Eddy covariance			9.3	Roulet et al., 2016
Canada	Ontario	45.68	-75.8	Temperate bog	Autochamber			9.5, 11.6	Lai et al., 2014
Canada	Ontario	45.68	-75.8	Temperate bog	Static chamber			4.9	Roulet et al., 2007
Canada	Ontario	45.68	-75.8	Temperate bog	Eddy covariance	19.5			Brown et al., 2014
Canada	Quebec	54.8	-66.82	Boreal fen	Static chamber	0.1			Moore and Knowles, 1990

Canada	Quebec	54.8	-66.82	Boreal fen	Static chamber	13.1			Moore and Knowles, 1990
Canada	Quebec	54.8	-66.82	Boreal rich fen	Static chamber	4			Moore and Knowles, 1990
Canada	Alberta	54.82	-112.47	Boreal fen	Eddy covariance	12.4			Long et al., 2010
Canada	Newfoundland	48.26	-58.67	Boreal bog	Chamber	1.7			Luan and Wu, 2015
Canada	Newfoundland	48.26	-58.67	Boreal bog	Eddy covariance	1.9, 2.0	1.7, 1.03	3.6, 3.1	This study

3.5. Discussion

3.5.1. Diel variation in CH_4 flux and its controls

We observed a clear diel pattern of CH_4 flux during the growing season, with an increase during the daytime and a sharp peak at around mid-day during 2014 and a broader peak between 13:00 and 18:00 in 2015. Similar diel patterns in CH_4 flux with higher daytime emissions than nighttime ones have been observed in some wetlands (e.g., a moderately rich treed fen in Alberta, Canada (Long et al., 2010), a boreal fen in Saskatchewan, Canada (Suyker et al., 1996) and a phragmites-dominated marsh in Nebraska, USA (Kim et al., 1999), following the diel trend of solar radiation, air/peat temperature, humidity or u* (Kim et al., 1999; Long et al., 2010; Suyker et al., 1996; Whiting and Chanton, 1993). We found that the diel pattern of CH_4 was mostly related to the diel variation of PPFD in 2014 and T_1 in 2015 at this bog. Nevertheless, the possible mechanisms by which PPFD and temperature regulate the diel pattern of CH_4 flux were previously reported to be similar, including plant-mediated bulk transport via convective flow (Knapp and Yavitt, 1992; Long et al., 2010; Morrissey et al., 1993), a stomatal pathway for the release of CH_4 (Garbet et al., 2005; Knapp and Yavitt, 1992; Long et al., 2010; Morrissey et al., 1993; Whiting and Chanton, 1996), and root exudation (Minoda and Kimura, 1994; Whiting and Chanton, 1996). Convective flow happens when pressure gradients from different parts of plants form due to the air-leaf temperature and humidity gradients that can be regulated by either temperature or PPFD, especially when vascular plants are present (Dacey, 1981). This mechanism may occur at this bog since it is comprised of hummock and hollow patches which are dominated by vascular species (Luan and Wu, 2014). For the mechanism of diffusion that occurs when gas concentration gradients exist, high PPFD or temperature increases the stomatal conductance (Knapp and Yavitt, 1992; Morrissey et al., 1993), which can promote the diffusion

of CH_4 out of leaves in plants based on laboratory experiment (Garnet et al., 2005). However, under field conditions no consistent results have achieved about whether movement of CH_4 from the leaf interior to the atmosphere is the rate-limiting step or not. Schimel (1995) suggested that stomata did impose the largest resistance to CH_4 diffusion and so could contribute to diel variation in CH_4 emission, while Chanton and Dacey (1991) indicated that resistance to diffusion of CH_4 between the soil and root would be the most important control on movement of CH_4 through plants. For the root exudation mechanism, although we found the peak time of CH_4 flux lagged the maximum PPFD time by an average of ~1h during the growing season of both 2014 and 2015 (Fig.S3.1.), we are not confident that the short time-lag can support the increased CH_4 production and subsequent diffusion. Overall, the diel pattern of CH_4 flux is controlled by complex processes, with several mechanisms operating at one time and EC measurements alone cannot sort this out.

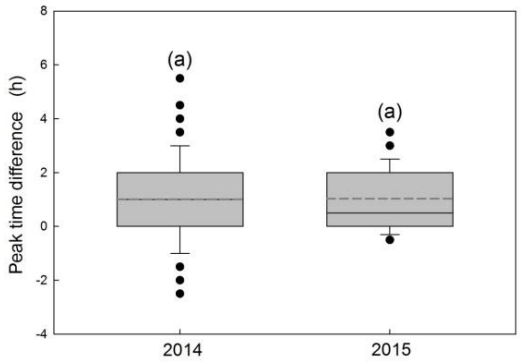


Fig.S3.1. Box plot of the peak time difference between photosynthetic photon flux density (PPFD) and CH_4 flux during the growing season in 2014 and 2015. Positive values indicate the peak time of CH_4 flux lags behind that of PPFD. The dashed grey line and solid black line indicate the mean and median values, respectively. The same letters above the error bar indicate the difference between variables is not significant.

3.5.2. Seasonal variation in daily CH_4 flux and its controls

Water table (Moore and Roulet, 1993), air and soil temperature (MacDonald et al., 1998; Moore and Dalva, 1993), substrate availability (Christensen et al., 2003) and their interactions have been shown to play an important role in regulating the growing season variations in CH_4 flux of peatlands. We found that soil temperature was most strongly linked to the temporal variation in growing season CH_4 flux in both 2014 and 2015, which is consistent with previous studies showing that soil temperature is the dominant control over CH_4 flux in undisturbed peatlands (Turetsky et al. 2014). The strong temperature control arises because of its direct impact on the processes of methanogenesis and CH_4 production, which are regulated by the activity of methanogenic bacteria (Metje and Frenzel, 2007). The positive relationship between CH_4 production and temperature has been well established in peatlands (Svensson, 1984; Williams and Crawford, 1984). Moreover, we found the seasonal variation in CH_4 flux was more dependent on T_{50} than T_1, probably resulting from the fact that CH_4 production occurs at subsurface peats below the water table. At this bog, the average growing season WT was -21 cm in 2014 and -23 cm in 2015. We examined the influence of soil temperature at 30 cm (near the average WT) on CH_4 flux, but its relationship was weaker than that of T_{50} (not shown).

WT was found to be an insignificant factor in regulating CH_4 flux in both years though it exerted some positive influence on growing season CH_4 flux in 2014 (Table 3.3) due to the non-linear relationship between them (Fig.3.8a, 3.8b). We found a slight increase in CH_4 flux as WT increased from -0.30 ~ -0.35 m to -0.20 ~ -0.25 m in both years, and then it significantly increased and decreased as WT approached the peat surface in 2014 and 2015 growing season, respectively. The different responses of CH_4 flux to WT when WT was close to the peat surface

may arise from different hydrology conditions in the warm growing season period in the two study years, with more frequent rises of water table in 2014 than in 2015 (Fig.3.3d). Therefore, in 2014 a large portion of high WT values available in this analysis were in the warm August and September, while shallow WT values mostly came from the cold early growing season in 2015 (Fig.3.3d). Indeed, we found that WT had a positive effect on CH_4 flux in warm period with T_{50} above 13 ^oC (Fig.3.8c) and a negative effect in cold period with T_{50} below 5 ^oC (Fig.3.8d) at this bog, implying that only a wet climate in warm growing season periods can promote the CH_4 flux in this boreal bog. The negative impact of WT on CH_4 flux in cold periods may be due to the overriding effect of low subsurface soil temperature, which limited CH_4 production due to low activity of methanogenic bacteria and reduced substrate availability (Turetsky et al., 2014). The regression analysis results suggested that PPFD and air/soil temperature were important controls over CH_4 flux during the early and late growing season in both years rather than WT (Table 3.3). Moreover, although we found the highest CH_4 emission rates occurred when WT was near its growing season mean value (-0.23 m) in 2015, we failed to observe any obvious on/off switch functioning of WT as was found at Mer Bleue bog (Brown et al., 2014). This was probably because of the higher mean and limited range in water table fluctuations at this boreal bog of -0.21~-0.23 m in the two study years than that of -0.40 ~ -0.50 m at Mer Bleue bog.

Substrate availability has been recognized as important control over CH_4 fluxes in this bog (Luan and Wu, 2015), similar to the findings in northern peatlands elsewhere (Alm et al., 1997; Bellisario et al., 1999; Christensen et al., 2003, Pypker et al., 2013, Waddington and Roulet, 1996; Whiting and Chanton, 1993). However, the results of our multiple regression analysis indicated that NEE was not a main controller over the temporal variability of CH_4 flux. This may

be due to different functions of NEE on CH_4 flux under varying daily CO_2 fixation capacity, where the average daily CH_4 fluxes increased with enhanced daily CO_2 uptake rate when daily net CO_2 uptake occurred, while no correlation between NEE and CH_4 flux existed when there is little or no CO_2 fixation (Fig. 3.9). This result is consistent with the finding from Pypker et al. (2013). During the growing season, the increase in plant productivity inputs high quality organic matter into peat soils (Luan and Wu, 2015), thereby facilitating the anaerobic CH_4 production and hence CH_4 emission. We found the highest CH_4 flux lagged the maximum NEE by 30-40 days (not shown). Similarly, Lai et al. (2014) observed that the daily average CH_4 flux lagged 18-26 days behind gross ecosystem production at Mer Bleue bog in Canada. These results further suggest that the CH_4 production potential may have been regulated by substrate availability from fresh OM input for this bog, as suggested by others (Liu et al., 2014). Overall, our results indicate that the growing season CH_4 fluxes were controlled by both abiotic factors (soil temperature and WT) and biotic factors (NEE).

During the non-growing season, the daily average CH_4 flux was mostly correlated to u* in both years (Table 3.3). The relationship between u* and CH_4 flux has been reported in several previous studies of tundra ecosystems and lakes (Sachs et al., 2008; Wille et al., 2008). Increased turbulence leads to a transient flushing of CH_4 that was stored in near surface layers during calm periods in wintertime, leading to high CH_4 fluxes.

3.5.3. Comparison of the annual CH_4 fluxes

On the annual time scale, this boreal bog acted as a small source of CH_4, with emissions of 3.58 $\pm$ 0.69 g CH_4 m^{-2} yr^{-1} in 2014-15 and 3.07 $\pm$ 0.67 g CH_4 m^{-2} yr^{-1} in 2015-16 (Table 3.4). These values are within the lower range of the reported CH_4 rates in undisturbed peatlands elsewhere

(Table 3.5). In addition, growing season accumulated CH_4 fluxes of 1.88 ± 0.66 g CH_4 m^{-2} in 2014 and 2.04 ± 0.56 g CH_4 m^{-2} in 2015 were similar to the estimate of 1.8 g CH_4 m^{-2} in 2013 based on chamber measurements (Luan and Wu, 2015). The non-growing season cumulative CH_4 emissions of 1.70 ± 0.20g CH_4 m^{-2} in 2014-15 and 1.03 ± 0.37 g CH_4 m^{-2} in 2015-16 accounted for 33 and 47% of the total annual emissions, similar to that of 43-46% in an alpine wetland (Song et al., 2015) and 35% in a subarctic peatland (Jackowicz-Korczyński et al., 2010). Clearly, non-growing season emissions cannot be ignored and highlights the need for year-round CH_4 measurements when quantifying the annual CH_4 budget. Although fluxes during the non-growing season are generally quite small, less than 1/2 that of average growing season emission rates at this bog as well as in other peatlands (Jackowicz-Korczyński et al., 2010; Song et al., 2015), the long duration of winter in high latitude regions (~six months) leads to substantial accumulated losses. In some studies high emission rates were observed during soil freezing and thawing periods (Jackowicz-Korczyński et al., 2010). We did not find any evidence of spring-thaw or fall freezing induced CH_4 emission pulses. Nevertheless, our data indicate the importance of non-growing CH_4 flux in the annual CH_4 emission. Moreover, the data quality of non-growing season CH_4 flux measured by the open-path LI-7700, due to the frequent snowing and other unexpected weather condition, was found to be much lower than the growing season CH_4 flux data. Therefore, there is a critical need for long-term non-growing CH_4 flux measurement in order to better elucidate the controls on the non-growing season CH_4 emission in boreal peatlands and their contribution to the overall annual budget.

3.6. Conclusion

We have assessed the magnitude, patterns and environmental drivers of CH_4 fluxes at different time-scales in a boreal bog based on two years of ecosystem-scale measurements. Our study

suggested that several controls affected the ecosystem CH_4 exchange interactively, but these determinants shifted among different time-scales, suggesting that different parameterizations are required to model the CH_4 emissions at different time scales in the peatland CH_4 modeling. Moreover, it is of particular significance to consider how future precipitation changes especially during warm growing season period to accurately predict the CH_4 flux in the future since we found the effect of WT on CH_4 was obvious and meaningful only in warm period. Although we found the diel pattern of CH_4 flux was closely related to PPFD in 2014 and T_1 in 2015, the exact mechanism controlling the diel pattern of CH_4 flux during the growing season is not well understood, which needs more process-level mechanistic experiments to address. In particular, this study highlights the importance of non-growing season CH_4 emissions in the estimation of annual CH_4 budget and bias may exist in estimating non-growing season CH_4 emissions from empirical relationships only based on growing season data since the controls over CH_4 flux can be different between growing season and non-growing season.

Chapter 4 - Near-zero methane emission from an abandoned boreal peatland pasture based on eddy covariance measurements

4.1. Abstract

Although a few estimates of the annual methane (CH_4) flux from agriculturally managed peatlands exist, knowledge of controls over the variation of CH_4 at different time-scales is limited due to the lack of high temporal-resolution data. Here we present CH_4 fluxes measured from May 2014 to April 2016 using the eddy covariance technique at an abandoned peatland pasture in western Newfoundland, Canada. The goals of the study were to identify the controls over the seasonal variations in CH_4 flux and to quantify the annual CH_4 flux of the abandoned peatland pasture. The seasonal variation in daily CH_4 flux was not strong in the two study years, however a few periods of pronounced emissions occurred in wintertime (2014-15) and the late growing season (2015-16). The daily average CH_4 flux was small relative to other studies, ranging from -4.1 to 9.9 nmol m^{-2} s^{-1} in 2014-15 and from -7.1 to 12.1 nmol m^{-2} s^{-1} in 2015-16. Results of both path analysis and stepwise multivariable analysis indicated that CH_4 flux was only weakly related to either water table depth or soil temperature. The annual CH_4 emission was near zero in both years (0.36 ± 0.30 g CH_4 m^{-2} yr^{-1} in 2014-15 and 0.13 ± 0.38 g CH_4 m^{-2} yr^{-1} in 2015-16), but fell within the range of CH_4 emissions reported for peatland pastures elsewhere.

4.2. Introduction

Agricultural drainage is one of the most common management practices in northern peatlands. About 20% of pristine peatlands have been drained for agriculture, forestry, and peat extraction, among which agriculture is now the most widespread human use for peatlands globally (Alm et

al., 2007; Frolking et al., 2011; Lohila et al., 2004; Maljanen et al., 2010; Oleszczuk et al., 2008). Although natural peatlands tend to be carbon dioxide (CO_2) sinks and methane (CH_4) sources, they have acted to cool global climate for the past several millennia, sequestrating ~20-30 g C m^{-2} yr^{-1} from the atmosphere, mainly due to slow decomposition rates of peat organic matter under waterlogged conditions (Wang et al., 2014; Yu et al., 2010). Agricultural drainage leads to significant alterations of the hydrology and vegetation of peatlands (Kopp et al., 2013), which can potentially affect their C cycle and their corresponding impact on climate (Luan and Wu, 2015). However, the importance of managed peatlands in global CH_4 cycling and climate regulation remains uncertain mainly due to the lack of knowledge of CH_4 flux processes and the underlying mechanisms, which requires reliable high-frequency CH_4 flux data to resolve (Kroon et al., 2010).

CH_4 has a significant climate warming potential, about 25 times that of CO_2 on a 100-year time horizon, and variations in the CH_4 flux can exert a great impact on regional and global climate (Pachauri et al., 2014). In peatlands, CH_4 is produced by methanogenic bacteria in the anaerobic layer and is emitted into the atmosphere through diffusion, ebullition and via plant aerenchyma (Lai, 2009). Ebullition and plant transport are relatively direct paths to the atmosphere, whereas CH_4 that diffuses through the soil matrix can be partly oxidized to CO_2 by methanotrophs in the overlying aerobic layer, reducing the flux considerably (Lai, 2009). Hence, the dynamics of the CH_4 flux are determined by the joint effects of the complex and changing processes of CH_4 production, consumption, and transport, which can vary with many factors, such as water table depth, soil water content, temperature, nutrient availability, vegetation composition, pH, redox potential, and physicochemical properties of soils (Adji et al., 2014; Goodrich et al., 2015; Lai,

2009; Le Mer and Roger, 2001; Pypker et al., 2013). As a result, CH_4 fluxes usually show great temporal and spatial variability (Koebsch et al., 2015; Kroon et al., 2010; Salm et al., 2009; Schäfer et al., 2012). A recent review suggested that water table depth and temperature are the dominant controls on CH_4 flux for pristine bogs and swamps, but their effects can be partly offset or even overridden by other processes such as vascular plant transport in some wetland types (Turetsky et al., 2014).

Drainage for agriculture can inhibit the release of CH_4 from peatlands by decreasing the thickness of the potential CH_4 production zone and increasing the thickness of the potential CH_4 oxidization zone. It can also favor the growth of aerenchymous plants enabling direct transport of CH_4 from the soil to the atmosphere (Schäfer et al., 2012), thus promoting CH_4 emissions. Although, in general, agricultural drainage has been suggested to decrease CH_4 emission (Schrier-Uijl et al., 2010), knowledge of the dynamic pattern and magnitude of CH_4 flux for managed peatland systems is limited especially on short time scales such as hours to days due to a lack of high-frequency measurements. Most earlier studies on CH_4 flux in agriculturally managed peatlands have been based on weekly or biweekly chamber measurements in European countries such as Finland (Maljanen et al., 2010), Sweden (Kasimir Klemedtsson et al., 2009) and Norway (Grønlund et al., 2008; Kløve et al., 2010).

In Canada, approximately 170,000 km^2 peatlands have been drained and converted for agricultural use (Joosten, 2009; Oleszczuk *et al.*, 2008). Although agricultural drainage is the most important conversion activity for peatlands in Canada, little is known about the magnitude and pattern of CH_4 exchange in these peatlands. Here, we examine a data set of half-hourly eddy

covariance (EC) CH_4 flux measurements during the period from April, 2014 to June, 2016 at an abandoned peatland-converted pasture in western Newfoundland, Canada. The objectives of the study were: 1) to assess the seasonal variations in CH_4 flux, 2) to identify the controls over the temporal patterns of the CH_4 flux and 3) to quantify the annual CH_4 flux at this site.

4.3. Methods

4.3.1. Site description

The study site is an abandoned peatland pasture with an average peat depth of ~4 m located in the Robinsons pasture, Newfoundland, Canada (48.264 °N, 58.665 °W) (Fig.4.1). The climate is oceanic temperate with an annual temperature averaging 4.5 °C and yearly rainfall of 1340 mm based on the previous 30 years' measurements from the nearest weather station 50 km from the site. The pasture (~ 0.2 km^2) was formerly a boreal bog that was drained by ditches in the 1970s and pasture forage grasses were introduced 35 years ago. The ditches were excavated to a depth of ~0.5 m and the width of ~30 cm along an east-west transect with a distance of 20-30 m between ditches. The site was used as pasture for 10 years and then abandoned. After the abandonment, the site was left to regenerate for ~25 years, but with continued active drainage (Luan and Wu, 2014; Luan and Wu, 2015). In its present state, the abandoned peatland pasture is dominated by perennial grasses and shrubs, but composed of several types of vegetation patches dominated by different species: patches dominated by reed canary grass (*Phalaris arundinacea)* and lower herbaceous and graminoid species (*Carex* spp., *Ranunculus acris*, *Ranunculus repens*, *Hieracium* sp.), and clusters of low shrubs, including sweet gale (*Myrica gale*), labrador tea (*Rhododendron groenlandicum*), mountain fly honeysuckle (*Lonicera villosa*), rhodora (*Rhododendron canadense*), and chokeberry (*Photinia* sp.). Despite this complex mix, the site is homogeneous in terms of vegetation composition with no obvious spatial pattern in vegetation

patches within the footprint of the EC tower. Plant characteristics were measured in a separate study in 2013, where peak aboveground biomass ranged from 225 to 591 g m^{-2} and root biomass varied from 186 to 340 g m^{-2} among different patches (Luan and Wu, 2014).

Fig.4.1 The location of flux tower in the Robinsons pasture, western Newfoundland, Canada (48.264 N, 58.665 W). The image was from Google Earth with imagery collected on May 28, 2006, where the outline of the site was indicated by the red solid line and the red pin represents the location of eddy covariance (EC) tower (a); (b) a photo of the setup of EC measurement system.

4.3.2. Flux and meteorological measurements

The CH_4 EC system consisted of a three-dimensional sonic anemometer (Gill WindMaster, Gill Instruments Ltd, Lymington, Hampshire, UK) to measure the vertical and horizontal wind vectors, and an open path infrared gas analyzer (LI-7700, LI-COR Inc., Nebraska, USA) to measure CH_4 concentration (Fig.4.1). The LI-7700 and anemometer were mounted at a height of 3.6 m from the ground surface, with the northward, eastward and vertical separation from sonic anemometer of 18 cm, -1cm, and 10 cm, respectively. Data output from the EC system were

recorded at 10 Hz with a data logger (LI-7550, LI-COR Inc., Nebraska, USA) and stored on a removable USB.

A set of meteorological instruments mounted on the EC system tower were used to continuously monitor environmental factors. Two quantum sensors (LI-190SL-50, LI-COR Inc., Nebraska, USA) were mounted to measure the photosynthetically active photon flux density (PPFD), with the upper one measuring the incoming PPFD and the lower one the reflected PPFD. Air temperature (Ta) and relative humidity (RH) were measured with air temperature and humidity probes, which were installed within a ventilated radiation shield (HMP155, Vaisala, Vantaa, Finland). A tipping-bucket rain gauge mounted on the ground was used to measure total event rainfall recorded at 30-min intervals (TR-525USW, Texas Electronics, Texas, USA). Soil temperature (Ts) was measured at 1 cm, 5 cm, 10cm, 30 cm, 50 cm, and 100 cm (LI7900-180, LI-COR Inc., Nebraska, USA) and soil moisture was measured as volumetric water content at 5 cm, 10 cm, 30 cm and 50 cm below the peat surface (Delta-TML2x, Delta-T Devices, Cambridge, UK). Water table (WT) was monitored by a stainless steel transducer pressure sensor with SDI-12/RS232 connection (CS451, Campbell Scientific, Utah, USA). A four-way net radiometer was mounted at 3.6 m height to measure incoming and reflected short-wave solar radiation and incoming and emitted long-wave radiation (CNR4, Kipp & Zonen, Delft, the Netherlands). All meteorological sensors, except for the rain gauge, were scanned at 5-s intervals and recorded as half-hourly means by a data logger (CR3000-XT, Campbell Scientific, Utah, USA) located in an insulated, heated and air-conditioned instrument hut.

4.3.3. Data Processing

EddyPro 5.2.1 software (LI-COR, Lincoln, NE, USA) was used to process the 10 Hz raw data

and output the corrected CH_4 flux over a 30-min interval. We used the default settings for statistical tests for raw high-frequency data (despiking) (Vickers and Mahrt, 1997), block averaging detrending, correction for frequency response [analytic high-pass filtering correction: (Moncrieff et al., 2004); low-pass filtering correction, select and configure: (Moncrieff et al., 1997)], density fluctuations (Webb et al., 1980), sonic anemometer tilt correction with double rotation (Wilczak et al., 2001), angle-of-attack correction for wind components (Nakai and Shimoyama, 2012), lag minimization using maximum covariance with default lag of 0, and calculation of friction velocity (u*) using both along and cross wind shear. Footprint lengths were calculated following Kljun et al. (2004) and quality flags for the flux calculation were determined following Mauder and Foken (2011). For high/low pass filtering, the correction procedure is described in detail in the EddyPro manual (LI-COR Biosciences, 2016), which is briefly reiterated here. Both high-pass and low-pass filtering corrections included four steps: 1) estimation of the true cospectra using a modification of the Kaimal formulation (Kaimal et al., 1972), 2) determining the high/low-pass transfer function (HPTF, LPTF) which is specified by the superimposition of a set of transfer functions describing sources of high/low frequency losses, 3) estimating flux attenuation by "applying" the calculated HPTF/LPTF to the modeled flux cospectra, and 4) calculating a high/low-pass spectral correction factor. For quality control and flagging, a steady state test which compares the statistical parameters determined for the averaging period and for short intervals within this period and integral turbulence characteristics test which compares the measured parameters and the modeled ones was applied. The deviation (%) of both the steady state and integral turbulence characteristics of less than 30 indicates good data quality, between 30 and 100 moderate quality and larger than 100 bad quality. The diagnostic flag related to data quality were output, with the values of 0, 1, 2, representing data

with high, intermediate, and poor quality, respectively. More related details of quality controls can be found in Mauder and Foken (2006).

The outputted half-hourly fluxes were corrected for spectral attenuations, air density fluctuations and instrument-specific effects as mentioned above. The magnitude of such correction factors were 1.06 and 1.12 in the growing season, 1.08 and 2.31 in the freezing period, 1.02 and 1.05 in the thawing period and 0.02 and 0.92 in the wintertime of the two study years. The poor quality data of CH_4 flux with quality flag of 2 and the mean value of received signal strength indicator (RSSI) for LI-7700 less than 20% were discarded. Fetch for the pasture site varies from about 170-370 m in different directions (0 -45 °: 200 m; 45-77 °: 287 m; 77-115 °: 370 m; 115-160 °: 170 m; 160-250 °: 360 m; 250-360 °: 200 m), so we discarded the flux data with the 70% cumulative footprint larger than these fetches. The footprints were mostly within 200 m during the different periods of both measurement years, but the dominant wind directions showed some differences among different periods (Fig.4.2). The dominant wind direction was from NNW to NNE during all seasons (Fig.4.2). We did not find a correlation between CH_4 flux and u*, thus failing to determine a u* threshold. Therefore, we set it as 0.1 m s^{-1} as in a previous study where no u* threshold could be found (Reichstein et al., 2005). Flux data with u* below 0.1 m s^{-1} were discarded in this study. The final flux data were corrected by adding the storage flux value below the height of the EC instruments. The storage flux is estimated from temporal changes in gas concentrations based on concentrations from the LI-7700 and the height integral between the instrument and peatland surface (LI-COR Biosciences, 2016), under the assumption that CH_4 concentrations were invariant with height. Our data (not shown) indicate that the CH_4 flux

storage was not highly variable and was one or two magnitude less than the corresponding flux values.

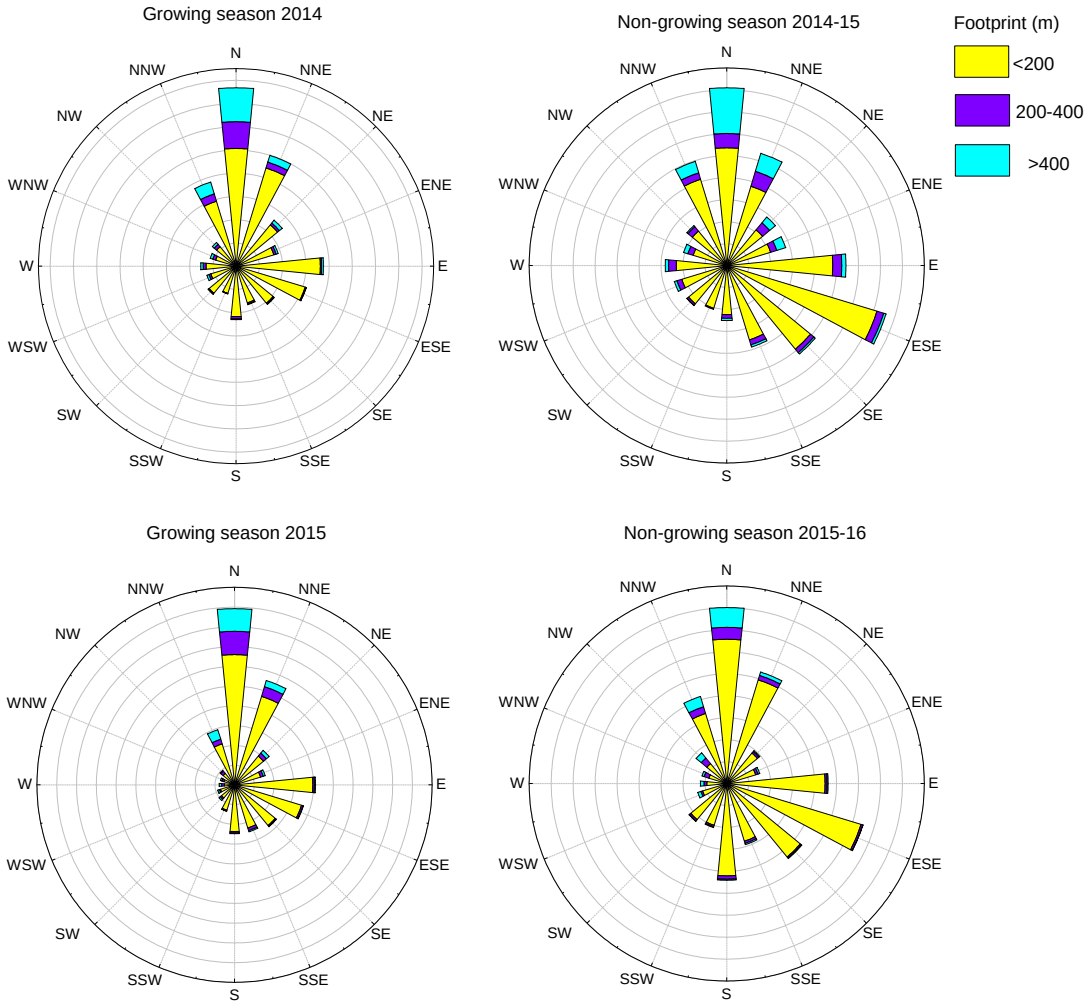


Fig.4.2. Footprint versus wind direction for different periods in the two study years. The legends indicate the cumulative footprint where 70% flux were originated. The yellow, purple and blue boxes indicate varying distances from the tower where the 70% of CH_4 fluxes were originated.

We divided the data into growing season and non-growing season as Song et al. (2015). The purpose of this division was to estimate the contribution of cumulative CH_4 flux in each period to the annual flux budget as well as to examine the variations in the controlling factors of CH_4 flux

in growing season and non-growing season. We further divided non-growing season into soil thawing, soil freezing and winter to investigate whether large CH_4 burst existed or not in the soil thawing and freezing period. The start and end of the growing season were determined as the dates where daily GPP exceeded 5% of the seasonal amplitude. In this study we assumed growing season began and ended after the first seven consecutive days with daily air temperature above 5°C and below 5°C, respectively. We also divided the growing season into three periods of early growing season (May and June), peak growing season (July and August) and late growing season (September, October and November). Soil freezing ranged from the end of the growing season to the first two consecutive days with average daily soil temperature below 0°C at 10 cm depth. Winter started at the end of the soil freezing period and ended when snow started melting (after seven consecutive days with average air temperature above 0°C). The soil thawing period was between the end of the winter period and the start of the growing season.

Most of the CH_4 flux data gaps were caused by power failures in extremely harsh weather and equipment failures, which resulted in a loss of 29% of the total flux record between May 2014 and April 2016. In addition, CH_4 flux data were discarded due to quality control, below u* threshold and footprint filtering, thus causing additional data gaps. Overall, during the growing season data gaps of CH_4 flux accounted for 43% and 35% of the total record in 2014 and 2015, respectively. During the non-growing periods 58% and 68% of the data were missing in 2014/15 and 2015/16, respectively. Currently, there is no consensus on gap-filling methods for CH_4 flux data and many different methods were used (Dengel et al., 2013; Forbrich et al., 2011; Kroon et al., 2010; Long et al., 2010; Parmentier et al., 2011; Rinne et al., 2007; Song et al., 2015; Sturtevant et al., 2012). Here we employed an artificial neural network (ANN) as proposed by

Dengel et al. (2013). We used the neural network Fitting Tool in the mathematical software Matlab to select data, create and train the network, and evaluate its performance using mean square error and regression analysis. Neural networks included an input layer, a hidden layer and an output layer (Elizondo and Góngora, 2005; Jain et al., 1996) and this two-layer feed-forward network with sigmoid hidden neurons and linear output neurons can fit multi-dimensional mapping problems arbitrarily well. Data were randomly divided into three sets: 70% of all data for training, 15% for testing and 15% for validating. Training data were presented to the network during training and the network was adjusted according to its error; validation data were used to measure network generalization, and to halt training when generalization stopped improving; testing data had no effect on training and so provided an independent measure of network performance during and after training. The network was trained with a Levenberg-Marquard back-propagation algorithm (trainlm) as used in previous studies (Dengel et al., 2013; Riedmiller, 1994). We chose input variables including air temperature, surface soil temperature, subsurface soil temperature, PPFD, vapor pressure deficit (VPD), u* and WT according to Dengel et al. (2013). However, during some period in wintertime, VPD and u* data were also missing, so we only used the remaining variables mentioned above to fill the data gaps at these times. To set a reliable number of neurons in the hidden layer, we applied 1-10 neurons to standardized approaches (Järvi et al., 2012). The training distribution showed a constant increase in correlation coefficient with increase in the number of neurons. Therefore, we set the number of neurons in the fitting network's hidden layer as 10. This procedure was replicated for 20 times and the median predictions were used to fill missing half-hour fluxes. Before training, all data were normalized between 0-1 (Aubinet et al., 2012; Dengel et al., 2013; Moffat et al., 2010; Nguyen and Chan, 2004) and divided into nighttime and daytime data sets according to a PPFD

threshold of 20 μmol m^{-2} s^{-1}. The gap-filled data were only used to calculate the total CH_4 flux during each period. All analyses presented below used measured data only, except for seasonal and annual totals of CH_4 flux, which were gap-filled. Fluxes away from the surface (i.e. CH_4 emissions) were treated as positive and fluxes into the surface (i.e. CH_4 sinks) were negative.

4.3.4. Uncertainty estimation

Although there are many uncertainty sources in flux estimation measured by eddy covariance, here we focused on flux random uncertainty due to sampling errors, and the flux uncertainty due to the gap-filling. The other uncertainty sources can be avoided due to either properly field experiment design (Businger, 1986) or data processing correction, thus sampling error will remain as one of the largest sources of uncertainty. Flux random uncertainty (σ_1) due to sampling errors is calculated following Finklestien and Sims (2001) in EddyPro and the details can be found in (Finkelstein and Sims, 2001). We estimated the flux uncertainty due to gap-filling (σ_2) based on the following procedures. Firstly, we developed, trained and validated ANN model using the available measured data in each study period (i.e., growing season, soil freezing period, soil thawing period and wintertime). Secondly, we ran the ANN model and produced a continuous series of data for the whole two-year study period. Finally, we compared the difference between the available measured data and their counterpart predicted CH_4 flux values from ANN model in each study period (Moffat et al., 2007). $\sigma_2 = 1 / N \sum (P_i - O_i)$. N means the number of available measured and predicted CH_4 flux pairs in each study period; P_i and O_i are the individual predicted CH_4 flux data and the observed flux value, respectively. The total uncertainty was calculated following the equation: $\sigma = [\sigma_1^2 + \sigma_2^2]^{1/2}$.

4.3.5. Statistical analyses

Pearson/exponential regression analysis and stepwise multivariable regression analysis were conducted to examine the effect of each environmental variable and their combined effects on CH_4 flux, including surface soil temperature at 1 cm (T_1), subsurface soil temperature at 50 cm (T_{50}), VPD WT, PPFD, u* and soil water content (SWC) at 30 cm. These analyses were conducted using the statistical program SAS v9.1. In addition, we applied path analysis which evaluates the causal connections between sets of variables on the basis of multiple regression to identify the relationships between explanatory variables affecting CH_4 flux using Amos.22 (SPSS AMOS, IBM). In path analysis, the decision coefficient is often used to quantify the integrated determination effect of environmental factor, containing not only the direct determination effect of a certain variable on the dependent variable, but also the indirect determination coefficient related to this variable (Huxman et al., 2003; Saito et al., 2009; Ueyama et al., 2014). Daily average values, based on the measured data with at least 70% of the data available for a single day, were used for the analysis and all data were normalized as 0-1 to approximately achieve a normal distribution before the analysis following the equation: Normalized values = (data- mindata) / (maxdata-mindata); where mindata and maxdata are the minimum and maximum value of each variable.

4.4. Results

4.4.1. Environmental conditions

The air temperature was close to the normal for most months during study periods (all values within one standard deviation of the respective 30-years means), with the exceptions of warmer conditions in July 2014 and September 2015 and colder than normal conditions in March, April, June, July and November 2015 and April 2016 (Table 4.1). Low rainfall in September 2015 was

notable, but higher than normal precipitation occurred in three consecutive winter months from November 2014 to January 2015 (Table 4.1).

Table 4.1 Comparison of monthly average temperature and cumulative monthly rainfall measured at Robinson Pasture during measurement periods from April, 2014 to May, 2016 with the long-term (1981-2010 average ± SD) measurements from the nearby, climate station in Stephenville, Newfoundland and Labrador.

Month	Rainfall (mm)				Air temperature (℃)			
	2014	2015	2016	1981-2010	2014	2015	2016	1981-2010
Jan		54	14	29 ± 24		-6.8	-5.3	-6 ± 1.6
Feb		20	45	27 ± 30		-9.2	-3.9	-6.7 ± 2.9
Mar		12	30	37 ± 29		-6.9	-5.0	-3.5 ± 2.5
Apr		41	88	62 ± 42	1.6	-0.8	0.6	2.6 ± 1.8
May	129	118	106	94 ± 44	6.5	7.1	7.4	7.6 ± 1.4
Jun	65	64		104 ± 45	12.2	10.2		12.1 ± 1.3
Jul	97	119		118 ± 45	19.0	14.1		16.4 ± 1.1
Aug	105	125		130 ± 65	16.5	17.9		16.7 ± 0.9
Sep	83	55		128 ± 48	12.2	13.7		12.8 ± 1.1
Oct	85	101		124 ± 45	8.5	6.4		7.4 ± 1.3
Nov	133	82		94 ± 31	1.5	1.2		2.7 ± 1.3
Dec	105	54		49 ± 42	-1.5	-2.1		-2.4 ± 1.7
Overall		845		995 ± 133		3.7		5.0 ± 1

Environmental variables for the two study years showed obvious seasonal patterns (Fig.4.3). The daily average air temperature ranged from ~-14.9 ℃ to 23.2 ℃ in the first study year and from ~ -11 ℃ to ~21 ℃ during the second study year, and the lowest values of both years occurred during middle-late February, while air temperature peaked in early July in 2014 and near the middle of August in 2015 (Fig.4.3: a1-a3). The daily average surface soil temperature at 1 cm ranged from ~-1.6 ℃ to ~20.1 ℃ in the first study year and from ~-1.6 ℃ to 19.5 ℃ in the second year, with the lowest values for both years occurring near the end of December when the freezing periods ended and winter period began. The highest values coincided with the peak in air temperature in each year (Fig.4.3: b1-b3). For subsurface soil temperature at 50 cm, the seasonal trend for both

years was quite similar, except with the peak delayed by 20 days in 2014 and 10 days in 2015 compared to the peak of soil surface temperature at 1 cm (Fig.4.3: b1-b3). The daily cumulative rainfall ranged from 0 mm to 89 mm in the first study year and from 0 mm to 53 mm in the second study year (Fig.4.3: f1-f3). Soil water content at 30 cm remained in a narrow range between 0.61 and 0.67 m^3 m^{-3} in both study years (Fig.4.3: d1-d3). Water table was always below the peatland surface in the first year, ranging from -61 to -3 cm, with a mean value of -28.4 cm, and ranged from -52 to 2 cm in the second year with a mean of -28.4 cm, when it was slightly above the peatland surface only in April 2016 (Fig.4.3: e1-e3, Table 2). Soil moisture and WT were high in the non-growing season and decreased to minimum values during the mid-growing season although both variables showed periodic sharp rises and decreases corresponding to summer rain events greater than 10 mm and the subsequent drawdowns (Fig.4.3: e1-e3). Mean growing season water table positions for the two years were -41.6 cm and -29.4 cm for 2014 and 2015, respectively (Table 4.2).

Table 4.2 Average air temperature (T_a), soil temperature (T_s) at depth of 1 cm and 50 cm, photosynthetic photon flux density (PPFD), cumulative rainfall, and water table for four different periods. Negative values indicate water table depth (WT) was below the peatland surface.

Period	Date	T_a	T_s (℃)		PPFD	Rainfall	WT
		(℃)	1 cm	50 cm	(mol m^{-2} d^{-1})	(mm)	(cm)
Growing season	2014.5.15-11.11	12.8	13.1	10.5	28.7	513	-41.6
	2015.5.16-11.15	11.4	11.8	9.9	25.4	603	-29.4
Soil freezing	2014.11.12-12.28	-0.4	0.9	5.8	4.9	176	-11.5
	2015.11.16-12.29	-1	1.1	5.4	5	92	-10.9
Winter	2014.12.29-2015.5.3	-5.8	0.3	2.4	18.2	126	-17.1
	2015.12.30-2016.5.1	-3.4	0	2.3	15.1	174	-11.1
Soil thawing	2014.5.1-5.14	2.6	5.8	3.3	35.1	112	-19.6
	2015.5.2-5.15	5.7	4.7	2.4	29.2	38	-7.7
Overall	2014.5-2015.5	4.3	6.9	6.7	22.7	936	-28.4
	2015.5-2016.5	4.6	6.3	6.5	20	890	-20.5

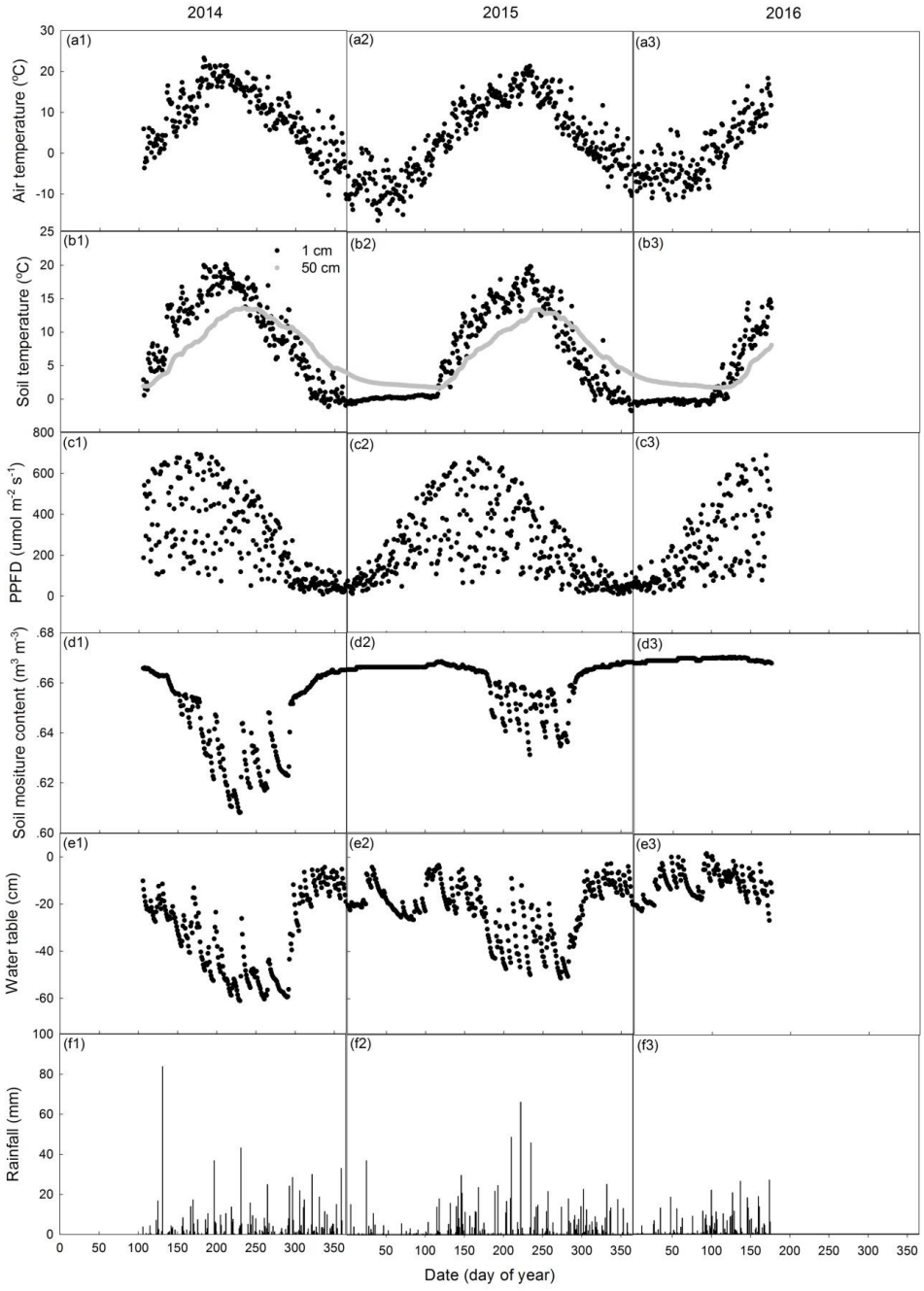


Fig. 4.3 The daily average air temperature (a1-a3), soil temperature at 1 cm and 50 cm (b1-b3), photosynthetic photon flux density (PPFD) (c1-c3), volumetric soil water content at depth of 30 cm (d1-d3), water table (e1-e3) and cumulative rainfall (f1-f3) during the measurement periods

4.4.2. Seasonal dynamics of CH_4 fluxes

The seasonal pattern of CH_4 fluxes was not strong in either study year, though some pronounced periods of emissions occurred in 2014/2015 winter and late growing season in 2015, but uptake occurred in all seasons in both years (Fig.4.4). In general, the CH_4 fluxes were small varying around zero, with the daily average CH_4 flux ranging from -4.1 to 9.9 nmol m^{-2} s^{-1} over the first study year and from -7.1 to 12.1 nmol m^{-2} s^{-1} over the second study year (Fig.4.4). The range of wintertime CH_4 emission fluxes was comparable to that of the growing season and CH_4 uptakes occurred during soil freezing period in both years (Table 4.3).

Although our annual CH_4 flux estimates suggested this abandoned pasture was a net source of CH_4 to the atmosphere, the annual totals were not significantly different from zero at 0.36 ± 0.30 g CH_4 m^{-2} yr^{-1} in 2014-15 and 0.13 ± 0.38 g CH_4 m^{-2} yr^{-1} in 2015-16 (Fig.4.5, Table 4.3). The largest uncertainty in the annual estimates came from random errors of 0.30 g CH_4 m^{-2} yr^{-1} in 2014-15 and 0.38 g CH_4 m^{-2} yr^{-1} in 2015-16 (Table 4.3). The flux bias associated with the gap-filling was neglected during the growing season in both years since the agreement between modeled and measured CH_4 fluxes was high (i.e., model efficiency >80%) (Table 4.3). However, during the non-growing season, the model efficiency was low of 20% due to the lack of strong dependence of CH_4 flux on environmental variables. As a result, the uncertainty due to the gap-filling was pronounced during wintertime of both study years and soil thawing period in 2014-15, with the bias accounting for 12% -50% of the flux budget of each period (Table 4.3).

Path analysis showed expected strong correlations between certain environmental variables, such as the positive correlation between air temperature and soil temperature at 1 cm, or between

PPFD or air temperature and VPD (Fig.4.6a, b). However, the only significant correlations with CH_4 flux were WT in 2014 and subsurface soil temperature at 50 cm in 2015 (Fig.6a, 6b), similar to the results of stepwise multivariable analysis and linear and nonlinear regression analysis (Table 4.4, 4.5). Regression analysis results indicated that the variation in the average daily CH_4 flux during the growing season was positively but weakly related to T_{50} in 2015 ($R^2 = 0.2$, $P < 0.0001$), but not in 2014 (Table 4.5). The correlation between daily average CH_4 flux and WT was positive but weak in 2014 ($R^2 = 0.1$, $P = 0.05$) but negative in 2015, and SWC exerted a negative impact on the daily average CH_4 flux in 2015 ($R^2 = 0.13$, $P = 0.001$) (Table 4.5). The variation in daily average CH_4 flux was not significantly related with u*, PPFD and VPD during the growing season in both 2014 and 2015 (Table 4.5).

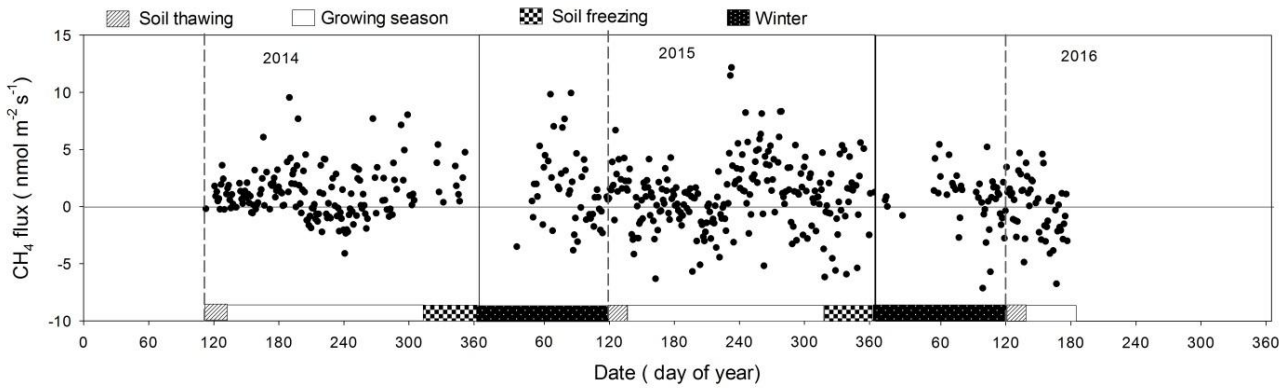


Fig.4.4 The daily average CH_4 flux of different periods of during the two study years.

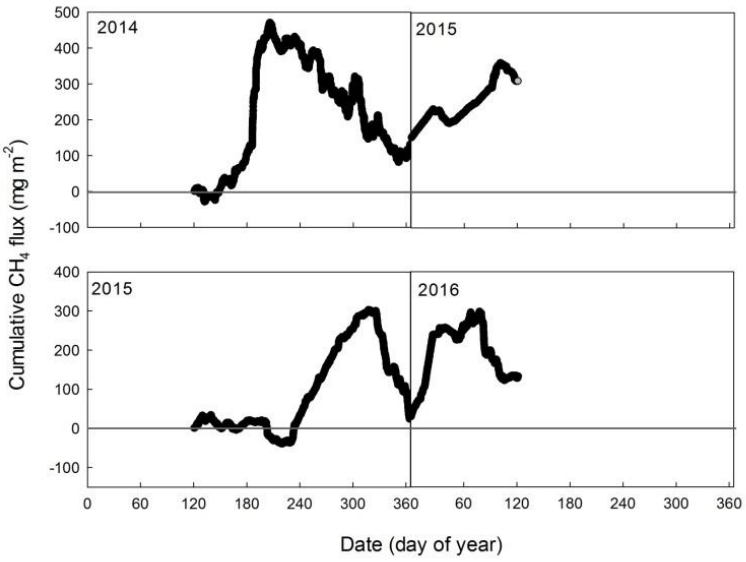
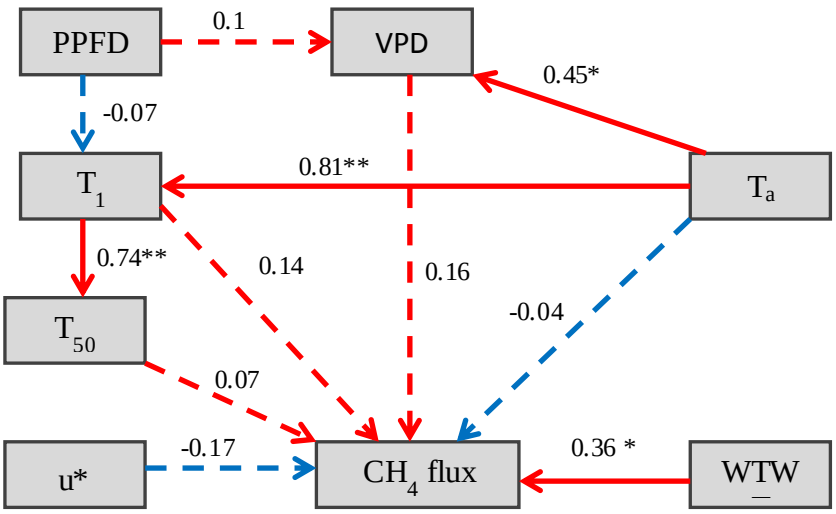


Fig.4.5 The cumulative gap-filled CH_4 flux during the two study years (from May 2014 to April 2016).

(a)



(b)

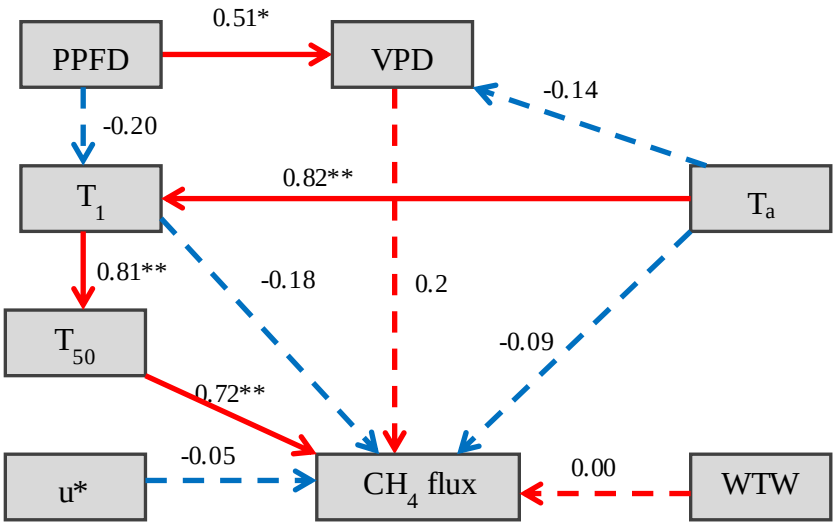


Fig.4.6 Path diagrams illustrating the relationship among environmental variables and CH_4 flux during growing seasons of 2014 (a) and 2015(b). Red solid arrows mean significant positive effect and blue arrows mean significant negative effect; the dashed arrows mean no significant correlation between variables. Numbers besides arrows are the standardized path coefficients and the width of lines indicates the strength of relationship.

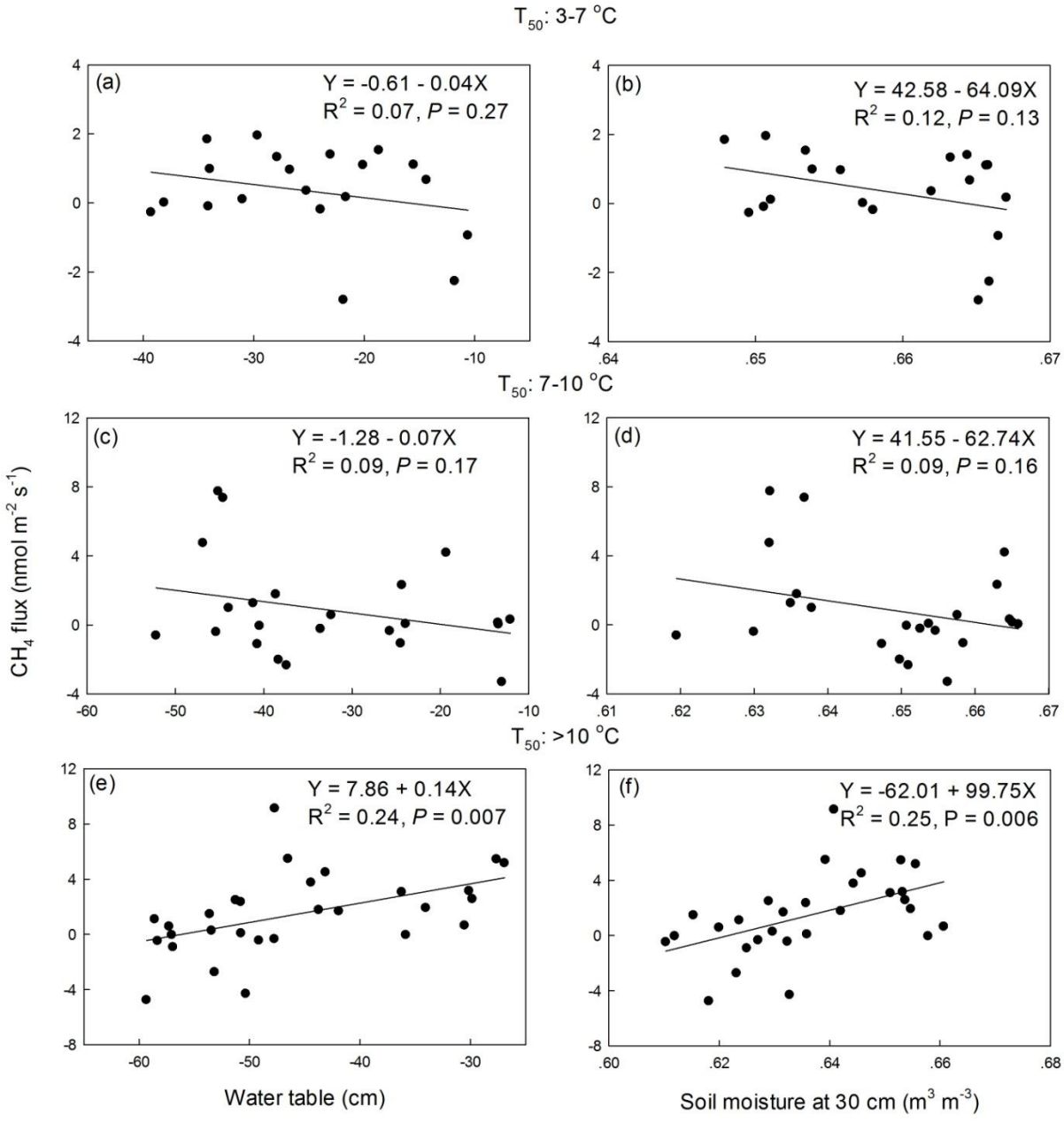


Fig.4.7 Relationship between CH_4 flux and water table and soil moisture content at 30 cm under different temperature condition. The relationships were based on daily averages; only days with more than 70% data coverage were used.

Table 4.3 Total accumulated CH_4 fluxes, their uncertainties (g CH_4 m^{-2}) for the different study periods and contributions to the annual emissions in two years from May 2014 to April 2016. RU, GU and TU in the table indicate random uncertainty, uncertainty due to gap filling and total uncertainty, respectively

Period	From May 2014 to April 2015							From May 2015 to April 2016						
	Duration days	CH_4 fluxes	Ratio of RU to flux	RU	Ratio of GU to flux	GU	TU	Duration days	CH_4 fluxes	Ratio of RU to flux	RU	Ratio of GU to flux	GU	TU
Growing season	181	0.17	1.49	0.25	0.02	0.003	0.25	184	0.27	1.15	0.32	0.03	0.01	0.32
Soil freezing	47	-0.04	0.59	0.02	0.05	0.002	0.02	44	-0.27	0.7	0.19	0.02	0.05	0.19
Winter	125	0.25	0.65	0.16	0.12	0.03	0.16	124	0.1	0.75	0.08	0.3	0.03	0.08
Soil thawing	12	-0.02	2.11	0.04	0.53	0.01	0.04	13	0.03	1.35	0.04	0.08	0.002	0.04
Annual Total	365	0.36	2.73	0.3	0.09	0.03	0.3	365	0.13	2.05	0.38	0.24	0.03	0.38

Table 4.4. The results of stepwise multivariable regression analysis between daily average CH_4 flux and environmental variables. Only days with more than 70% data availability were used in the regression analysis. Only significant ($P<0.05$) variables were included in the equation.

Year	Equation	R^2	P	df
2014	$Y = 0.45 + 0.26WT$	0.1	0.04	62
2015	$Y = 0.53 + 0.40T_{50}$	0.20	<0.0001	79

Table 4.5. The results of linear and nonlinear relationship between the daily average CH_4 flux and the related environmental variables during the growing season in 2014 and 2015. Only days with more than 70% data availability were used in the regression analysis.

Variable	Year	R^2	*P*	Equation
u*	2014	0.04	0.09	$Y = 0.49 - 0.15X$
	2015	0.01	0.31	$Y = 0.33 - 0.11X$
VPD	2014	0.02	0.28	$Y = 0.40 + 0.10X$
	2015	0.03	0.1	$Y = 0.35 - 0.13X$
PPFD	2014	0.01	0.42	$Y = 0.42 + 0.06X$
	2015	0.01	0.58	$Y = 0.31 + 0.0X - 0.18X^2$
T_a	2014	3.01E-06	0.99	$Y = 0.45 - 0.001X$
	2015	0.01	0.43	$Y = 0.25 + 0.07X$
T_1	2014	0.01	0.70	$Y = 0.40 + 0.23X - 0.23X^2$
	2015	0.06	0.09	$Y = 0.38 - 0.58X + 0.63X^2$
T_{50}	2014	0.07	0.12	$Y = 0.39 + 0.40\ X - 0.40X^2$
	2015	0.2	<0.0001	$Y = 0.11e^{1.41X}$
SWC_{30}	2014	0.09	0.06	$Y = 0.32 + 0.55X - 0.43X^2$
	2015	0.13	0.001	$Y = 0.48 - 0.28X$
WT	2014	0.07	0.05	$Y = 0.40 + 0.13X$
	2015	0.08	0.01	$Y = 0.39 - 0.20X$

Table 4.6 Comparison of accumulated methane flux balance for agriculturally managed peatlands.

Location		Latitude	Longitude			CH_4 flux g C m^{-2}		
Country	City	°N	°E	Peatland type	Study Method	Growing season	Annual average	Ref.
Finland	Markku Lappalainen	62.67	30.83	Drained for grass	Chamber	-0.17	0.13	Nykanen et al, 1995
Finland	Jokioinen	60.82	23.5	Drained for grass	Chamber	-0.18~-0.08	-0.17~0.64	Maljanen et al., 2003a,b, 2004, 2009, 2010; Regina et al., 2007
Sweden	Västra Götaland	58.33	13.5	Drained for grass	Chamber	0.09	0.12	Kasimir et al. , 2009
Norway	Bodø	67.28	14.47	Drained for grass	Chamber		1.5~1.6	Grønlund et al., 2006; Kløve et al., 2010
Netherland	South Holland	52.03	4.77	Drained for grass	Eddy covariance & Chamber		14.6-20.3	Kroon et al., 2010; Schrier-Uijl et al., 2010
USA	California	38.1	-121.64	Drained for grass	Eddy covariance		11.4	Knox et al., 2015
Canada	Napierville	45.13	73.43	Drained for crop	Chamber	-0.06~-0.08	0.2	Glenn, 1993
Canada		48.26	58.67	Drained for grass	Chamber	1.1		Luan and Wu, 2015
Canada		48.26	58.67	Drained for grass	Eddy covariance	0.1-0.1	0.3-0.4	This study

4.5. Discussion

4.5.1. Comparison of long-term CH_4 flux with other peatland pastures

With a few exceptions CH_4 flux from managed peatlands has been considered to be insignificant for the greenhouse gas balances (Grønlund et al., 2006; Kasimir Klemedtsson et al., 2009; Kløve et al., 2010; Maljanen et al., 2003, 2004, 2010; Regina et al., 2007). Our study measured the total annual CH_4 emissions were small, not significantly different from zero in the second study year (0.36 ± 0.30 g CH_4 m^{-2} yr^{-1} in 2014-15 and 0.13 ± 0.38 g CH_4 m^{-2} yr^{-1} in 2015-16) during the two study years, similar to the range of values from its counterparts in European countries and Canada (-0.17 to 2.8 g CH_4 m^{-2} yr^{-1}), but lower than the 11.4 g CH_4 m^{-2} yr^{-1} observed in California, USA and the 14.6-20.3 g CH_4 m^{-2} yr^{-1} measured in the Netherlands (Table 4.6). In the latter two cases, the high CH_4 mission rates were attributed to relatively high temperatures throughout the year at the California site (Knox et al., 2015) and the continuous application of decomposable organic materials which improved the substrate for methane production at the Dutch pasture (Hendriks et al., 2010; Schäfer et al., 2012). Moreover, the low growing season CH_4 emission rates of ~0.2-0.3 g CH_4 m^{-2} we observed in 2014 and 2015 were similar to a growing season rate of ~1 g CH_4 m^{-2} based on chamber measurements at our site in 2013 (Luan and Wu, 2015), both of which are within the range of -0.18-1.1 g CH_4 m^{-2} per growing season measured elsewhere in managed peatlands (Table 4.6). Overall, we attribute the low emissions at this peatland pasture to the relatively deep aerobic layer resulting from the low water table, which averaged ~ -42 cm in 2014 and ~ -30 cm in 2015 during the growing season. We assume that CH_4 produced in anaerobic layers below the WT was largely oxidized before being emitting to the atmosphere, resulting in these extremely low emissions. CH_4 uptake was observed in all seasons at this site, which is not unusual in managed peatland systems. For example, growing

season CH_4 uptake was found at an intensively managed grass peatland in the Netherlands (Van den Pol-van Dasselaar et al., 1997) and at a fen drained and converted to grassland in Finland (Nykanen et al., 1995).

4.5.2. Controls on the seasonal dynamics of CH_4 flux

Unlike the strong positive effect of temperature on CH_4 flux in natural peatlands (Yvon-Durocher et al., 2014), we found that subsurface soil temperature at 50 cm exerted a weak but positive effect on growing season CH_4 flux in this abandoned peatland pasture in 2015 ($R^2 = 0.13$, $P = 0.001$), but had no significant impact in 2014 ($R^2 = 0.07$, $P = 0.12$). The lack of strong dependency of CH_4 flux on subsurface soil temperature was probably attributable to low WT in the warm periods of mid-summer, where WT mostly fell to 40 cm below peat surface in 2014 and 2015 (Fig.3e). When the water table was low, CH_4 production and oxidation may be similar in magnitude, or CH_4 oxidation dominated, at which time the effect of the CH_4-temperature relation may be masked. This principle has been confirmed in several previous studies (Daulat and Clymo, 1998; Moore and Dalva, 1993a; Strack and Waddington, 2007; Turetsky et al., 2008).

Our data did not show that water table position had a strong effect on CH_4 flux during the growing season of the both study years. The lack of a positive effect of water table on CH_4 flux during the growing season was probably a result of the high water table positions which tend to occur during the cold periods of the growing season, such as the early and late growing season when CH_4 emission may be limited by low temperature and substrate availability. Previous studies have also found that high water levels were usually associated with cool weather conditions leading to negative or weak correlations between WT and CH_4 flux because temperature is limiting the CH_4 flux at these times (Brown, et al., 2014; Noyce et al., 2014;

Turetsky et al. 2014). Indeed, we found WT and soil water content had no significant effect on CH_4 flux when subsurface soil temperature at 50 cm was less than 12 ^oC (Fig.4.7a,f), though a significant positive function of WT and SWC on the daily average CH_4 flux was found when soil temperature was above 12 ^oC (Fig.4.7g ,h).

4.6. Conclusion

This study updates our knowledge of the short-term variations of CH_4 flux and its controls during the growing season at an abandoned boreal peatland pasture based on high temporal-resolution CH_4 flux data. This study found the CH_4 flux of the abandoned peatland pasture was very low, not significant in the C balance estimation, consistent with previous findings in other agriculturally managed peatlands. In addition, we found neither distinct temporal (diel, seasonal) pattern of CH_4 flux nor any significant driver, causing significant challenges for modeling CH_4 flux in this environment.

Chapter 5 - Can the abandoned peatland pasture sequestrate more carbon dioxide than the adjacent boreal bog?

5.1. Abstract

Net ecosystem exchange of carbon dioxide (NEE) and its components gross primary productivity (GPP) and ecosystem respiration (ER) were compared between a bog and an abandoned peatland pasture within the same peatland complex in western Newfoundland, Canada. Measurements based on the eddy covariance technique from April 2014 to April 2016 were used to examine the influence of agricultural management and abandonment on peatland carbon exchange. NEE, GPP and ER at both sites showed pronounced seasonal variation, with the CO_2 uptake and emission rates increasing from the non-growing season to the growing season and peaking near the middle growing season. The maximum net CO_2 uptake rate of -28.61 μmol m^{-2} s^{-1} and emission rate of 26.01 μmol m^{-2} s^{-1} at the pasture were significantly higher than those of -9.97 μmol m^{-2} s^{-1} and 13.05 μmol m^{-2} s^{-1}, respectively, at the bog. Daytime average NEE was related to photosynthetic photon flux density (PPFD) and air temperature (T_a) and the nighttime average ER decreased with soil water content, but increased with air and soil temperature for both sites. The annual CO_2 uptake of the pasture, NEE of -85 ± 40 g C m^{-2} yr^{-1} in 2014-15 and -82 ± 37 g C m^{-2} yr^{-1} in 2015-16, was larger than that of the bog, which was -18 ± 14 g C m^{-2} yr^{-1} in 2014-15. GPP of 1106 ± 144 g C m^{-2} yr^{-1} in 2014-15 and 990 ±124 g C m^{-2} yr^{-1} in 2015-16 and ER of 1021 ± 138 g C m^{-2} yr^{-1} in 2014-15 and 908 ± 118 g C m^{-2} yr^{-1} in 2015-16 at the pasture were about 2.5 times the corresponding fluxes in the bog. The difference in GPP between the bog and pasture was mainly related to different aboveground biomass. Higher ER at the pasture was probably related to its lower water table, greater substrate availability and higher autotrophic respiration. Unlike

previous findings that managed peatlands are large CO_2 emitters, our results suggest that abandoned peatland pastures can function like natural grasslands and sequester considerable amounts of CO_2 from the atmosphere.

5.2. Introduction

Despite only covering 3% of the earth surface, the carbon (C) storage of northern peatlands accounts for approximately one-third of the world's soil C stock (Gorham, 1991; Turunen et al., 2002; Yu et al., 2010a). Cold, waterlogged conditions and *Sphagnum*-dominated recalcitrant litter in peatlands leads to low decomposition of peat organic matter (OM), benefiting C accumulation (Gorham, 1991). Approximately 20% of natural peatlands have been converted for agricultural cultivation worldwide (Couwenberg, 2011; Joosten and Clarke, 2002; Turetsky and Louis, 2006). Such management involves drainage and crop cultivation (Turetsky and Louis, 2006), which can alter the peatland C exchange making agriculturally managed peatlands "hotspots" for carbon dioxide (CO_2) and other greenhouse gas emissions (Grønlund et al., 2006, 2008; Kasimir-Klemedtsson et al., 1997; Lohila et al., 2004). On one hand, C emissions at agriculturally managed peatlands are promoted by two factors. Firstly, drainage and subsequent lowering of the water table level exposes more peat to aerobic conditions and accelerates the aerobic mineralization rate of peat (Laiho, 2006). As a result, the accumulated C is emitted to the atmosphere mainly as carbon dioxide (CO_2). Secondly, agriculturally managed peatlands are usually dominated by crop plants or forage grasses, which produce highly decomposable litter (Couwenberg, 2011; Grønlund et al., 2008; Minkkinen et al., 1999), accelerating the decomposition rate of the newly accumulated OM. On the other hand, agricultural management can increase the CO_2 uptake capacity of peatlands through cultivation of productive plants such as reed canary grass (Karki et al., 2016; Shurpali et al., 2009, 2010) and the addition of nutrients

(Järveoja et al., 2015).

It has been estimated that approximately 25.8 t CO_2-C ha^{-1} yr^{-1} are lost from agriculturally managed peatlands globally (Couwenberg, 2011). However, this estimate is very uncertain since it is based largely on a limited number of peat subsidence measurements where the C loss is estimated from peat shrinkage, peat bulk density and C content (Couwenberg, 2011). Subsidence rates are highly variable among different managed systems, ranging from 2 to 40 mm yr^{-1} (Grønlund et al., 2008; Hooijer et al., 2012; Kasimir-Klemedtsson et al., 1997; Leifeld et al., 2011; Lohila et al., 2004; Pronger et al., 2014) depending on the site type, pre-drainage peat thickness, the time after drainage and other factors (Hooijer et al., 2012; Pronger et al., 2014). In addition, peat subsidence results from not only the oxidation of peats, but also other processes such as soil compaction due to loss of supporting pore water pressure, wind and water erosion and leaching of soluble organic matter (Hooijer et al., 2012; Leifeld et al., 2011; Pronger et al., 2014). The percentage of C loss attributable to peat oxidation is variable, making it difficult to accurately determine the direct on-site C losses (Couwenberg, 2011; Leifeld et al., 2011; Page et al., 2011). Moreover, the subsidence method only estimates the mean C loss since drainage was initiated (which may be decades ago) and does not provide information on the contemporary and ongoing CO_2 exchange (Couwenberg, 2011; Krüger et al., 2015) preventing us from predicting their current role in the global C cycle.

The eddy covariance (EC) technique, which measures landscape-scale CO_2 exchanges between the ecosystem surface and the atmosphere with high temporal-resolution, has been applied to study the contemporary CO_2 fluxes for several managed peatlands (Fleischer et al., 2016;

Morrison et al., 2013; Mudge, 2009; Rutledge et al., 2015; Schrier-Uijl et al., 2010a; Veenendaal et al., 2007).

Earlier results suggested a large variability in CO_2 fluxes from these ecosystems with both losses and sinks of CO_2 reported, but most showing significant losses (Aslan-Sungur et al., 2016; Grønlund et al., 2008; Hargreaves et al., 2003; Karki et al., 2016; Knox et al., 2015; Lloyd, 2006; Maljanen et al., 2001, 2004, 2010; Matthes et al., 2015; Rutledge et al., 2015; Soussana et al., 2004; Vuichard et al., 2007). Combining this with an accounting for other greenhouse gases (GHGs) and the effects of management practices, a recent synthesis study concluded that wetland conversion for agricultural management generally leads to a large positive radiative forcing on the Earth's climate (Petrescu et al., 2015). However, such studies have almost exclusively considered active agricultural management, the effects of long-term abandonment after agricultural conversion is largely unknown.

In Canada, peatlands cover an area of approximately 1.136 million km^2, second only to those in Russia (Tarnocai et al., 2005). During the past century, extensive areas of Canadian peatlands have been drained for various purposes, such as agriculture, forestry, horticulture and other uses (Joosten, 2009). Agricultural management of peatlands is the most common type of non-harvesting use in Canada (Joosten, 2009), with an area of 170,000 km^2 having been converted for such use, accounting for 15% of the total national resource of peatlands and mires (Oleszczuk et al., 2008). The principal uses are vegetable and small fruit production and pasture (Stewart, 1977). An estimated 3.5 Mt CO_2 yr^{-1} is emitted from Canadian peatlands drained for agriculture (Joosten, 2009). As in other cases, noted above, this estimate was based on limited number of

subsidence measurements and is therefore quite uncertain. To the best of our knowledge, very few studies have measured the landscape-scale CO_2 fluxes from an agriculturally managed peatland system in Canada. The purpose of this study was to fill this knowledge gap by comparing the multi-year landscape-scale CO_2 fluxes from EC measurements between an undisturbed boreal bog and an adjacent abandoned peatland pasture, both belonging to the same peatland complex. Our main objectives were to quantify and compare multi-year net ecosystem exchange of CO_2 flux for these two systems and identify the controls over CO_2 flux at different temporal scales. In this experiment we considered the bog to be the control, or background case.

5.3. Materials and Methods

5.3.1. Study site

The research sites are located in the Robinsons pasture, western Newfoundland, Canada (48.264 º N, 58.665 º W). According to the data from the nearest (31 km away) weather station in Stephenville

(48.541 ºN, 58.55 ºW)

(http://climate.weather.gc.ca/climate_normals/results_1981_2010_e.html?stnID=6740&autofwd=1), the average annual air temperature and precipitation are 5.0 ± 1 °C (mean ± standard deviation) and 1340 ± 180 mm for the past 30 years [1981 to 2010], respectively. Mean annual precipitation is partitioned into 995 ± 133 mm as rainfall and 393 ± 89 mm as snowfall. For the growing period, May to October, the temperature and precipitation averaged ~12.2 ± 1.2 °C and ~705 ± 102 mm.

The sites are part of a peatland complex that comprises an undisturbed boreal bog and an

abandoned peatland pasture. The abandoned peatland pasture (~0.2 km^2) was originally an ombrotrophic bog that was drained in the 1970s by drainage ditches (~0.5 m in depth and ~30 cm in width, with a distance of 20-30 m between ditches) oriented along an east-west transect. At the time of initial drainage pasture forage grasses were introduced. The site was used as a manage pasture for ~10 years and then abandoned. After the abandonment, the site was left to regenerate for ~25 years, but with the drains intact. The bog (~0.36 km^2) was located immediately adjacent to the east of the abandoned peatland pasture (Fig.5.1). The bog is a typical peatland type in Newfoundland, with component landforms of hollows, hummocks and pools and a substrate dominated by brown bog moss species (*Sphagnum warnstorfii* and *Sphagnum capillifolium*) and partly with gray reindeer lichens (*Cladina* spp.). A high abundance of ericaceous shrubs and sedges grows in both hollows and hummocks, but hollows are dominated by sedges and hummocks by shrubs. The dominant species of sedge is *Trichophorum cespitosum* and the main species of ericaceous shrubs are *Gaylussacia* spp. and *R. groenlandicum*. In 2013, the dry aboveground biomass (AGB) in the hummocks was estimated at 197 ± 87 g m^{-2} and a similar amount (191 ± 41 g m^{-2}) was estimated for the hollows (Luan and Wu, 2015). The abandoned peatland pasture is a mosaic of patches, which are dominated by canary grass (*Phalaris arundinacea*), various low herbaceous and graminoid species (*Carex* spp., *Ranunculus acris*, *Ranunculus repens*, *Hieracium* sp.), and several dwarf shrubs [sweet gale (*Myrica gale*), labrador tea (*Rhododendron groenlandicum*), mountain fly honeysuckle (*Lonicera villosa*), rhodora (*Rhododendron canadense*), and chokeberry (*Photinia* sp.)]. Dry AGB of patches dominated by shrubs and reed canary grasses was 591 ± 246 g m^{-2}, significantly higher than that of the ditches (225 ± 50 g m^{-2}) (Luan and Wu, 2015).

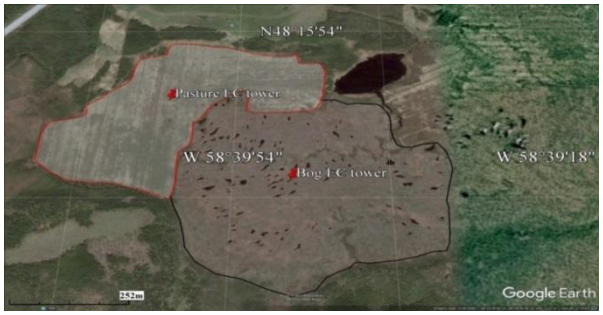
5.3.2. Eddy covariance flux and meteorological measurements

Two identical EC systems were operated for the period from April 2014 to August 2015 in the bog and from April 2014 to April 2016 in the abandoned peatland pasture (Fig.5.1). Data for the bog from the end of August in 2015 to April 2016 were unavailable due to equipment failure. Each system consisted of a three-dimensional (3-D) sonic anemometer (Gill WindMaster Pro, Gill Instruments) and a fast response infra-red gas analyzer (IRGA: LI-7200 Enclosed CO_2/H_2O Analyzer, Li-Cor Inc., Nebraska, USA). The 3-D sonic anemometers measured wind speed (u, v, w) and direction and sonic temperature at 3.44 m height for the bog and 3.7 m height for the abandoned peatland pasture. The LI-7200 analyzers were mounted at the height of 3.21 m for the bog and 3.54 m for the abandoned pasture to simultaneously measure variations in CO_2 and H_2O molar densities. Air was pulled by a diaphragm pump through a 1 m long sample tube to the IRGA at a rate of 16.1 L min^{-1} for the bog and 16.7 L min^{-1} for the abandoned pasture. In order to calculate the CO_2 and H_2O fluxes, instantaneous measurements of air temperature and air pressure inside the sampling cell were made. Two thermocouples were used to measure the instantaneous temperatures of air just before entering and immediately after leaving the sampling volume. A differential pressure sensor with a high speed and precision, together with a low speed, and a high precision absolute pressure sensor were used to measure instantaneous pressure in the middle of the cell. These data are required to calculate the mixing ratios (mole fractions) for CO_2 and H_2O. . Data output from the EC systems was recorded at 10 Hz with a data logging interface (LI-7550, Li-Cor Inc., Nebraska, USA) and stored on a removable USB device.

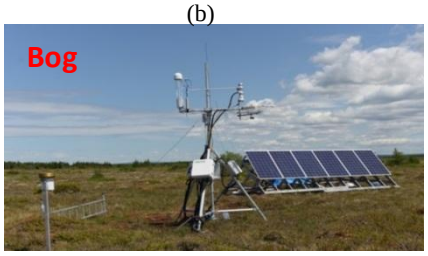
For both sites, a series of environmental variables were recorded by meteorological instruments mounted on the EC system tower. Photosynthetically active photon flux density (PPFD) was measured by q upward and downward facing quantum sensors (LI-190SL-50, LI-COR Inc.,

Nebraska, USA) and air temperature (T_a) and relative humidity (RH) were measured with temperature and humidity probes that were installed within a ventilated radiation shield (HMP155, Vaisala, Vantaa, Finland). Four-component net radiometers were used to monitor the incoming and reflected short-wave radiation and incoming and emitted long-wave radiation (CNR4, Kipp and Zonen, Delft, the Netherlands). A tipping-bucket rain gauge was mounted on the ground to measure total event rainfall (TR-525USW, Texas Electronics, Texas, USA). We measured the soil temperature at the depth of 1 cm, 5 cm, 10cm, 30 cm, 50 cm and 100 cm for the abandoned peatland pasture and at the same depths to 50 cm at the bog (LI7900-180, LI-COR Inc., Nebraska, USA). (LI7900-180, Li-Cor Inc., Nebraska, USA). Soil moisture was measured at the depth of 5 cm, 10 cm, 30 cm and 50 cm below peat surface for both sites (Delta-TML2x, Delta-T Devices, U.K.). Water table depth (WT) was monitored by a stainless steel transducer pressure sensor with SDI-12/RS232 connection (CS451, Campbell Scientific, Utah, USA). Rainfall was recorded at 30-min intervals and all the other environmental variables were scanned at 5-s intervals and recorded as half-hourly means by a data logger (CR3000-XT, Campbell Scientific, Utah, USA) located in an insulated, heated and air-conditioned instrument hut.

(a)



(b)



(c)

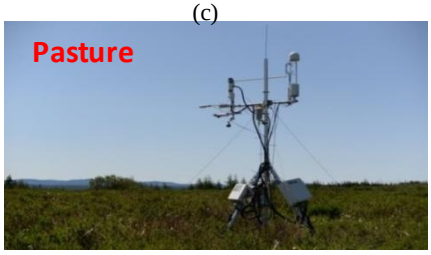


Fig.5.1. The location of flux towers at the bog and abandoned peatland pasture in the Robinsons pasture, western Newfoundland, Canada (bog: 48.260 °N, 58.663 °W; pasture: 48.264 °N, 58.665 °W).The image was from Google Earth with imagery collected on May 28, 2006, with the outline of the site was indicated by the red and black solid line and the red pin represents the location of eddy covariance (EC) tower (a); one photo of the setup of EC measurement system for the bog (b) and pasture (c).

5.3.3. Data Processing

Data quality control, and gap filling and partitioning

We used EddyPro 5.2.1 software (Li-Cor Inc., Nebraska, USA) to process the 10 Hz raw data and output the corrected fluxes of CO_2 and H_2O over a 30-min interval. We used the default settings for statistical tests for raw high-frequency data (despiking) (Vickers and Mahrt, 1997), block averaging detrending, correction for frequency response [analytic high-pass filtering correction: (Moncrieff et al., 2004); low-pass filtering correction, select and configure: (Moncrieff et al., 1997)], density fluctuations (conversion to mixing ratio) (Burba et al., 2012), sonic anemometer tilt correction with double rotation (Wilczak et al., 2001), angle-of-attack correction for wind components (Nakai and Shimoyama, 2012), lag minimization using maximum covariance with default lag of 0, and calculation of friction velocity (u*) using both along and cross wind shear. The WPL-terms correction was not applied to CO_2 and H_2O fluxes since the enclosed analyzer can output instantaneous mixing ratios of CO_2 and H_2O which are corrected for air expansion and contraction caused by the fluctuations in water, temperature, and pressure terms. Footprint lengths were calculated following Kljun et al. (2004) and quality flags were determined following Foken et al. (2004). The outputted corrected half-hourly fluxes have

diagnostic flags related to the data quality, with the values 0, 1, 2 representing data with good, moderate, and poor quality, respectively. The threshold of u* for CO_2 flux was determined as 0.12 m $s^{-1}$for the bog and 0.15 m s^{-1} for the pasture according to the online tool (http://www.bgc-jena.mpg.de/~MDIwork/eddyproc/). Poor quality data (flag of 2), u* less than 0.12 m s^{-1} for the bog and 0.15 m s^{-1} for the pasture, rainfall larger than 0, and negative nighttime NEE were discarded. The online tool was also used to do CO_2 gap filling and partitioning (http://www.bgc-jena.mpg.de/~MDIwork/eddyproc/). The gap-filling of the eddy covariance and meteorological data was performed through methods that are similar to Falge et al. (2001), but also consider both the co-variation of fluxes with meteorological variables and the temporal auto-correlation of the fluxes (Reichstein et al., 2005). The meteorological variables of Rg (global shortwave radiation input), T_a, T_1, RH, VPD and u* were needed for flux gap-filling. The method considers three different gap lengths with associated different gap-filling methods.1) If all meteorological data are available, the missing flux is filled by the average value under similar meteorological conditions with the difference of Rg, T_a and VPD less than 50 Wm^{-2}, 2.5 ^{0}C and 5.0 hPa, respectively within a time-window of 7 days. If no similar meteorological conditions are present within the time window, the averaging window is increased by 14 days. 2) If T_a or VPD data are missing, the flux gaps are filled using the same approach as case 1, but similar meteorological conditions are only defined by the deviation of Rg less than 50 Wm^{-2}. And 3) If all meteorological data are unavailable, the missing flux value is replaced by the average value at the same time of the day. The window size starts with 0.5 days and increases until the value can be filled. Nighttime data, when Rg was less than 20 Wm^{-2}, were defined as ecosystem respiration (ER). We used the Lloyd and Taylor (1994) regression model to fit the measured ER versus soil temperature at 1 cm as follows

$ER(T) = ER_{ref} * e^{E0[1/(Tref-T0)-1/(T-T0)]}$, where T_0 is constant at -46.02 ^{o}C [as in Lloyd and Taylor (1994)] and T_{ref} is set as 10 ^{o}C. E_0, the activation-energy parameter that determines the temperature sensitivity and is allowed to vary. ER_{ref} is the ecosystem respiration at reference temperature. Gross primary production (GPP) was calculated from ER and NEE: $GPP = ER - NEE$. Gap-filled flux data were only used for CO_2 budgeting in the following analyses.

5.3.4. Uncertainty estimation

Although there are many sources of uncertainty in eddy covariance fluxes, here we focused on the random uncertainty due to sampling errors. Other uncertainty sources such as errors due to the buoyancy effects of heat and water vapor, errors due to limited response time of the sensors, errors due to separation of the sensors and etc. can be avoided due to either careful field experimental design (Businger, 1986) or accounted for by flux data corrections as discussed above. Hence, sampling error is likely the largest unresolved source of uncertainty (Finkelstein and Sims, 2001). Measured flux random uncertainty (σ_1) is computed in the EddyPro software following Finkelstein and Sims (2001). This method requires the preliminary estimation of the Integral Turbulence time-Scales (ITS), which can be defined as the integral of the cross-correlation function between vertical wind component and any scalar of interest (e.g. temperature, gas concentration, etc.). Uncertainty due to gap-filling (σ_2) is also estimated by the EddyPro software and is based on the methods in (Falge et al., 2001; Reichstein et al., 2005) according to the following procedures. In brief, the method involves simulating gaps in the measured data and applying the gap-filling procedure described above. The error is computed from the difference between the measured data and their gap-filled counterpart as $\sigma_2 = 1 / N \sum (P_i - O_i)$, where N is the number of available measure (O_i) and predicted (P_i) flux pairs. The total uncertainty was calculated following the equation: $\sigma = [\sigma_1^2 + \sigma_2^2]^{1/2}$.

5.3.5. MODIS products and phenological indexes

We retrieved an index of vegetation phenology and greenness, which is related to the vegetation biomass from the MODIS satellite product (MOD13Q1:http://daacmodis.ornl.gov/cgibin/MODIS/GLBVIZ_1_Glb/modis_subset_order_global_col5.pl). The MOD13Q1 product provides the Enhanced Vegetation Index (EVI) at 250 m pixel resolution at 16-day intervals. The EVI product was selected because it has improved sensitivity over high biomass regions (Xiao et al., 2004). For the bog, the pixel of 250 m *250 m lies within the borders of the site, while about 1-2% of the pixel in pasture was beyond its edge and covers the edge of the bog. We also derived a set of phenological indices from daily GPP using the TIMESAT software package (Jönsson and Eklundh, 2004), including the start of the growing season (GSS), end of the growing season (GSE), length of the growing season (GSL), annual peak GPP rates and date of the peak GPP rates. The values of the start and end of the growing season were determined as the dates where daily GPP exceeded 5% of the seasonal amplitude. In this study we used the asymmetric Gaussian smoothing method embedded in TIMESAT, which has superior performance over other curve-fitting outputs (Hird and McDermid, 2009).

5.3.6. Data analysis

We analyzed the seasonal variation of photosynthetic efficiency response to light availability and the carbon uptake capacity during the growing season based on the following equation: $GPP = \alpha * GPP_{max} * PPFD / (GPP_{max} + \alpha * PPFD)$, where α is the initial slope of the light-GPP curve and is interpreted as the ecosystem light use efficiency (i.e., the carbon uptake per photon PPFD incident) and the modeled parameter GPP_{max} is the maximum carbon fixation rate at unlimited PPFD. These two parameters are variable in time and were calculated on a monthly basis. We

also calculated the dependency of ecosystem respiration on soil temperature and its sensitivity to temperature based on the following equations: $ER = \alpha * e^{\beta T}$ and $Q_{10} = e^{10\beta}$, respectively Q_{10} is a common index to indicate temperature sensitivity of soil decomposition, where larger Q_{10} values indicate greater sensitivity of organic matter respiration with increasing temperature (Liski et al., 1999). ER and T represented measured nighttime NEE and soil temperature at 1 cm, during the study period respectively; α and β are modeled parameters. These parameters were derived using the curve fitting tool in Matlab.

Linear/exponential regression analysis was conducted to examine the effect of SWC at 30 cm and T_1 on nighttime ER, and PPFD and T_a on daytime GPP for both sites. We also examined the difference in CO_2 fluxes between sites (ΔNEE, ΔER and ΔGPP), where the difference (Δ) was computed as pasture minus bog. Relationships between ΔNEE and its components (ΔER and ΔGPP) were studied using linear regression analysis. The relationships between the difference in EVI and WTD between sites (ΔEVI and ΔWTD) and ΔNEE, ΔER and ΔGPP were also analyzed using linear regression analysis. Since EVI is measured only once every 16 days, we averaged the C fluxes for the 16-day period centered on the EVI date (i.e., 8 days before and 8 days after). This procedure then generated a mean flux for each EVI date, which was then used in the analysis. These analyses were conducted using the statistical program SAS v9.1 and only days with more than 70% of the measured data available for both sites were used in the analysis.

5.4. Results

5.4.1 Environmental variables

Since the two research sites were located in the same peatland complex, they share similar T_a and precipitation regimes. For much of the study period T_a was near long-term (1981-2010) averages

(i.e., within ± 1 standard deviation) (Table 5.1). Anomalous periods include higher than normal T_a in July 2014 and September 2015 and colder than normal conditions in March, April, June, July and November 2015 and April 2016 (Table 5.1). Rainfall was lower than normal in September 2015, but higher than normal in three consecutive winter months from November 2014 to January 2015 (Table 5.1). Seasonal trends in environmental variables for the two study years showed expected patterns and were similar between sites (Fig.5.2). The daily average T_a was near zero before 20th April (day110) and increased above 5 ^{o}C after 15th May (day 135) in both years, peaking at late July in 2014 (day 183-212) and mid- to late-August (day 229-237) in 2015 and then dropped below 5 ^{o}C on 8th November (day 312) in 2014 and 15th November (day 319) in 2015. The highest T_a of 24 ℃ in 2014 was about 2 ℃ higher than that in 2015. The seasonal pattern of T_1 followed the seasonal variation in T_a and the highest values coincided with the peak in T_a in each year (Fig.5.2: b1-b3). The lowest values of T_1 for both years were reached near the end of December and remained near zero until the end of April the following spring. The daily average PPFD was highly variable during the growing season for both sites, ranging from less than 100 μmol m^{-2} s^{-1} to a maximum of about 720 μmol m^{-2} s^{-1} in May of 2014 and 680 μmol m^{-2} s^{-1} in June of 2015 (Fig.5.2f). Daily rainfall ranged from 0 mm to 89 mm in the first study year and from 0 mm to 53 mm in the second study year (Fig.5.2: e1-e3). Maximum WTs at the pasture site (-61 cm in 2014-15 and -52 cm in 2015-16) in were deeper than those at the bog (-33 cm in 2014-15 and -36 cm in 2015-16). Soil moisture at 30 cm remained in a narrow range between ~0.76 and 0.88 m^3 m^{-3} at the bog and between 0.61 and 0.67 m^3 m^{-3} at the pasture site in both study years (Fig.5.2: c1-c3). Soil moisture and WT were high in the non-growing season and decreased to minimum values during the mid-growing season, although both variables

showed periodic sharp rises and decreases corresponding to summer rain events greater than 10 mm and the subsequent drawdowns (Fig.5.2: d1-d3).

Table 5.1 Comparison of monthly average temperature and cumulative monthly rainfall measured at Robinson Pasture during measurement periods from May, 2014 to April, 2016 with the long-term (1981-2010 average ± SD) measurements from the nearby, climate station in Stephenville, Newfoundland and Labrador. Please note that bog and pasture had similar rainfall.

Month	Rainfall (mm)				Air temperature (℃)						
	Bog & Pasture				Bog	Pasture	Bog	Pasture	Bog	Pasture	
	2014	2015	2016	1981-2010	2014		2015		2016		1981-2010
Jan		54	14	29 ±24			-6.7	-6.8	-5.2	-5.3	-6 ±1.6
Feb		20	45	27 ±30			-9.2	-9.2	-3.9	-3.9	-6.7 ±2.9
Mar		12	30	37 ±29			-6.9	-6.9	-5	-5	-3.5 ±2.5
Apr		41	88	62 ±42			-0.8	-0.8	0.6	0.6	2.6 ±1.8
May	129	118		94 ±44	6.5	6.5	7	7.1			7.6 ±1.4
Jun	65	64		104 ±45	12.3	12.2	10.1	10.2			12.1 ±1.3
Jul	97	119		118 ±45	19.3	19	14.2	14.1			16.4 ±1.1
Aug	105	125		130 ±65	16.7	16.5	18.1	17.9			16.7 ±0.9
Sep	83	55		128 ±48	12.4	12.2	13.8	13.7			12.8 ±1.1
Oct	85	101		124 ±45	8.6	8.5	6.5	6.4			7.4 ±1.3
Nov	133	82		94 ±31	1.6	1.5	1.2	1.2			2.7 ±1.3
Dec	105	54		49 ±42	-1.4	-1.5	-2.1	-2.1			-2.4 ±1.7
Overall		845		995 ±133			3.7	3.7			5.0 ±1

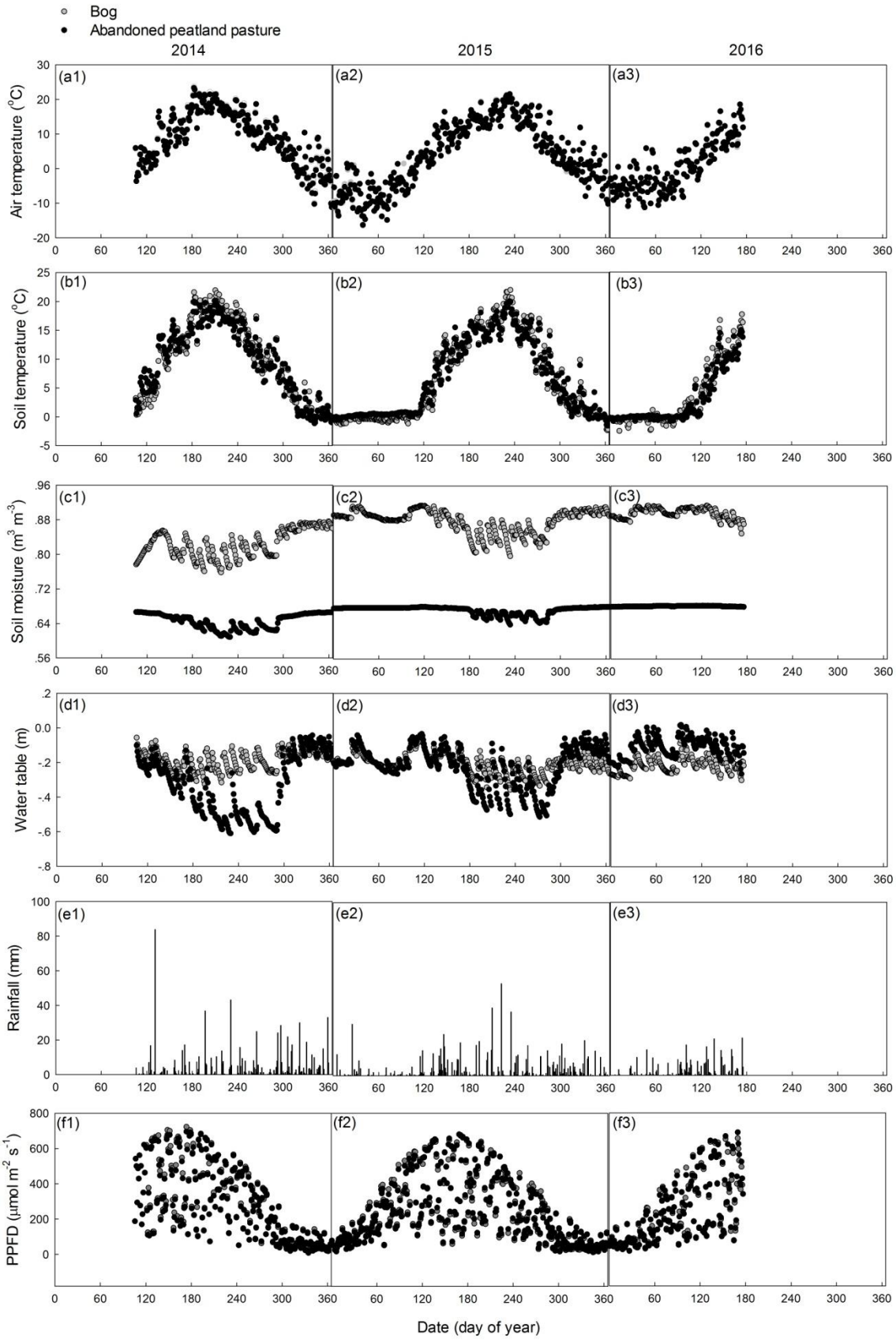

Bog
Abandoned peatland pasture
2014
2015
2016
(a1)
(a2)
(a3)
Air temperature (°C)
(b1)
(b2)
(b3)
Soil temperature (°C)
(c1)
(c2)
(c3)
Soil moisture (m³ m⁻³)
(d1)
(d2)
(d3)
Water table (m)
(e1)
(e2)
(e3)
Rainfall (mm)
(f1)
(f2)
(f3)
PPFD (μmol m⁻² s⁻¹)
Date (day of year)

Fig.5.2. The daily average air temperature (a1-a3), soil temperature at 1 cm (b1-b3), volumetric soil water content at depth of 30 cm (c1-c3), water table depth (d1-d3), cumulative rainfall (e1-e3) and photosynthetic photon flux density (PPFD) during the measurement periods for both bog and abandoned peatland pasture.

5.4.2. Comparison of the temporal variation in CO_2 flux

Although NEE displayed a strong diel pattern from May to October in both 2014 and 2015 for both sites, the abandoned peatland pasture had substantially larger half-hourly fluxes than at the bog (both daytime and nighttime values) during months from June to September (Fig.5.3). Only during the shoulder months of May and October were the diel magnitudes of NEE similar for both sites. Peak daytime CO_2 uptake and nighttime emission occurred in the warmest months (July and August) at both sites (Fig.5.3). NEE, ER and GPP showed strong seasonal patterns for both sites with significantly larger daily fluxes observed at the abandoned peatland pasture (Figs.5.4, 5.5). The daily average NEE ranged from -2.61 to 1.77 μmol m^{-2} s^{-1} in 2014-15 at the bog, and from -5.88 to 4.22 μmol m^{-2} s^{-1} in 2014-15 and from -6.38 to 4.12 μmol m^{-2} s^{-1} in 2015-16 at the pasture (Fig.5.4a, 5.5a). The highest daily net CO_2 uptake occurred in early August of 2014 at the bog, but in early July of both years at the pasture. ER ranged from 0.23 to 3.40 μmol m^{-2} s^{-1} in 2014-15 at the bog, and between 0.28 and 8.64 μmol m^{-2} s^{-1} in 2014-15 and between 0.20 and 8.43 μmol m^{-2} s^{-1} in 2015-16 at the pasture (Fig.5.4b, 5.5b). ER at both sites was low during the non-growing season (from November to April), generally less than 1 μmol m^{-2} s^{-1}, and increased steadily after snowmelt, peaking in the summer months. Peak daily ER fluxes of 8.64 μmol m^{-2} s^{-1} in early August of 2014 and 8.43 μmol m^{-2} s^{-1} in late August of 2015 at the pasture were significantly larger than maximum ER at the bog (3.40 μmol m^{-2} s^{-1}) which occurred in

mid- to late-July 2014. GPP followed a similar seasonal pattern as ER for both sites, with the highest daily average GPP of 3.70 μmol m^{-2} s^{-1} in 2014 at the bog and 10.17 μmol m^{-2} s^{-1} in 2014 and 9.75 μmol m^{-2} s^{-1} in 2015 at the pasture. The GPP peak occurred in early August for the bog and early- to mid-July in both years at the pasture (Fig.5.4c, 5.5c).

Monthly totals emphasized the much larger fluxes observed at the pasture compared to the bog during the growing season months, while the between-site differences were small during non-growing season months (Fig.5.6). The monthly mean GPP from June to September ranged from 4.91 to 10.02 μmol m^{-2} s^{-1} in 2014-15 and from 4.63 to 9.75 μmol m^{-2} s^{-1} in 2015-16 at the abandoned peatland pasture, whereas at the bog the range was 2.04-3.66 μmol m^{-2} s^{-1} in 2014-15 (Fig.5.6c). Monthly ER at the abandoned peatland pasture during June to September ranged from 4.27 to 6.97 μmol m^{-2} s^{-1} in 2014 and from 3.73 to 6.84 μmol m^{-2} s^{-1} in 2015 and was 1.52-2.72 μmol m^{-2} s^{-1} at the bog in 2014 (Fig.5.6b). Net CO_2 sinks occurred in June to September in both years at the abandoned peatland pasture, NEE ranged from -0.64 to -3.05 μmol m^{-2} s^{-1} in 2014 and from -0.20 to -3.78 μmol m^{-2} s^{-1} in 2015, and in 2014 at the bog NEE ranged from -0.27 to -1.91 μmol m^{-2} s^{-1} (Fig.5.6a).

Ecosystem light use efficiency of the pasture was significantly larger than that at the bog in June and July of 2014, but was similar at the two sites from August onward. In 2015, α was larger at the pasture from June to August (Fig.5.7a). Conversely, GPP_{max} of the abandoned peatland pasture was significantly larger than that of the bog in all growing season months in both years, except May when GPP_{max} was different between the two sites in 2014, but not in 2015. These contrasting results for May were attributed to a delay of ~15 days in the start of growing season

in 2015 for the abandoned peatland pasture compared to that in 2014 (Fig.5.7b).

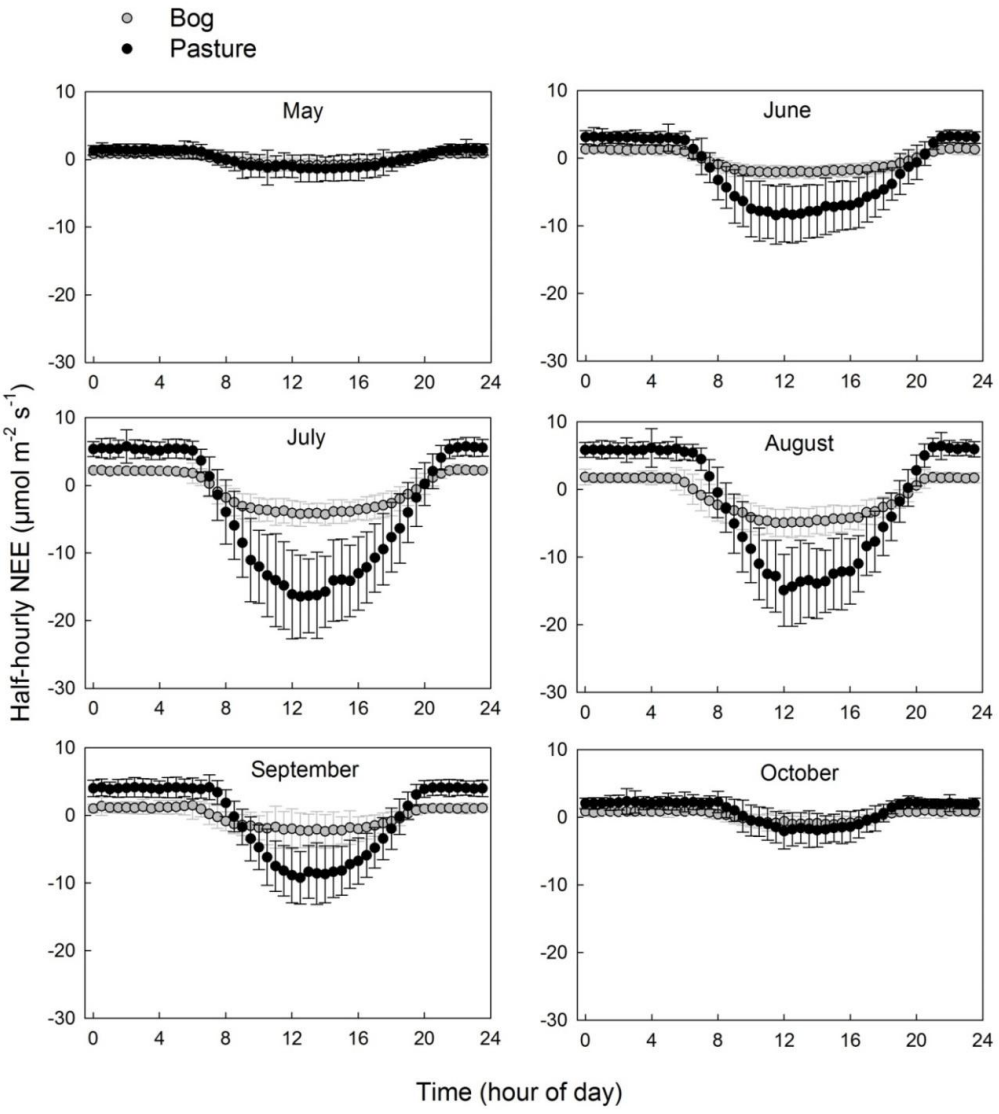


Fig. 5.3 Comparison of average diel variation in net ecosystem exchange of CO_2 (NEE) in different growing season months for the bog and abandoned peatland pasture sites. Data points are average values for all available measurements in each 30-minute period of the month and error bar indicates standard error of the average.

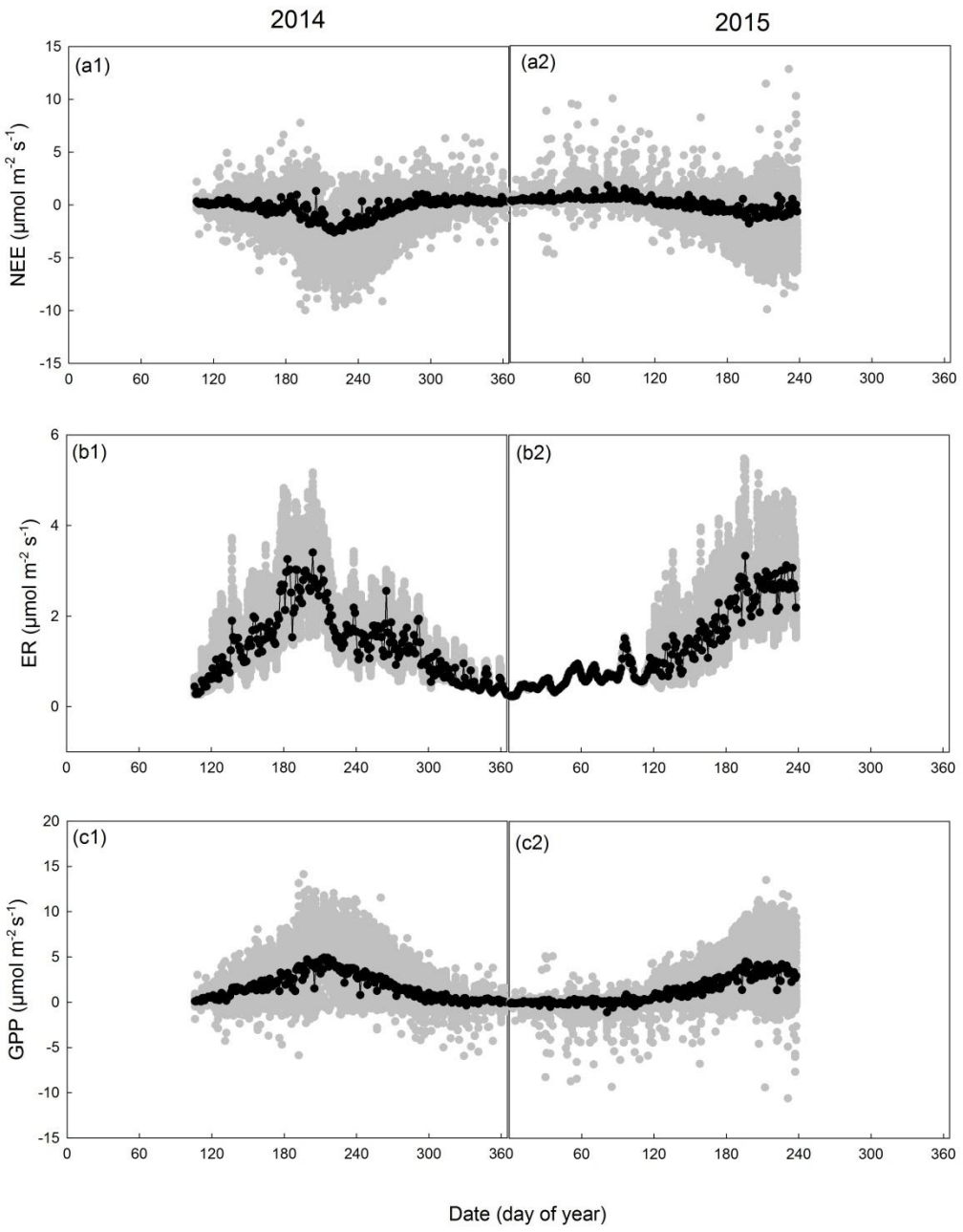


Fig. 5.4. Seasonal variation of half-hourly (gray dots) and daily average (black dots) NEE (a1-a3), ER (b1-b3) and GPP (c1-c3) during the period between April 2014 and August 2015 at the bog.

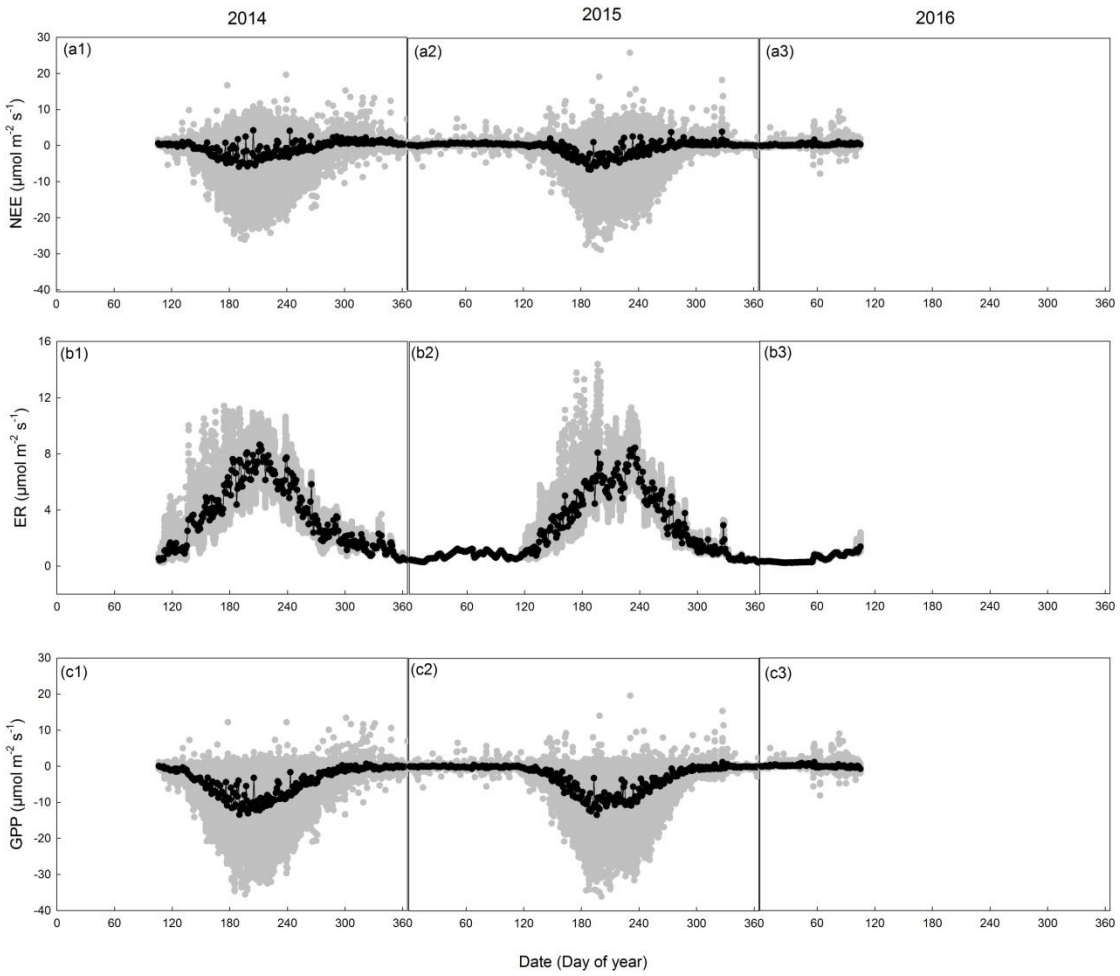


Fig. 5.5. Seasonal variation of half-hourly (gray dots) and daily average (black dots) NEE (a1-a3), ER (b1-b3) and GPP (c1-c3) during the period between April 2014 to April 2016 at the abandoned peatland pasture.

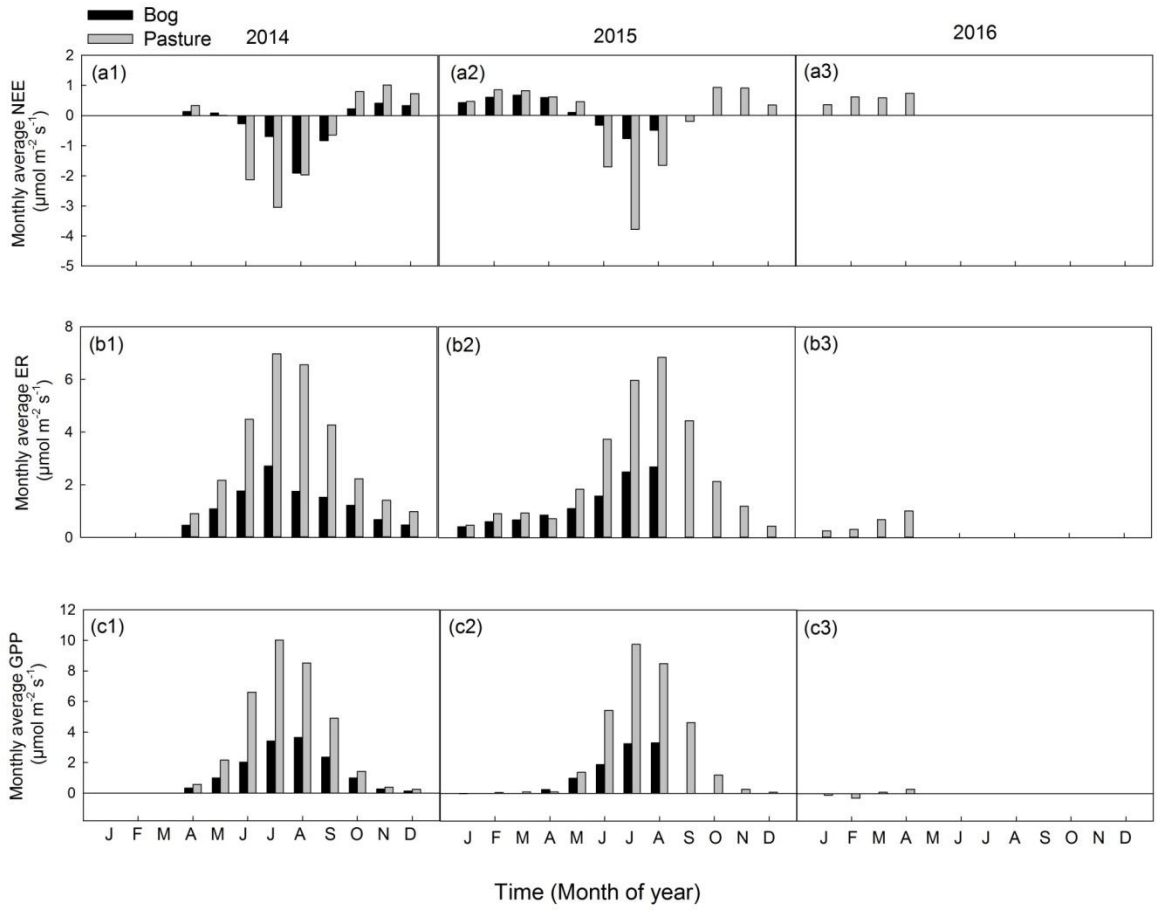


Fig.5.6 Monthly average NEE and its component fluxes GPP and ER for the bog and abandoned peatland pasture.

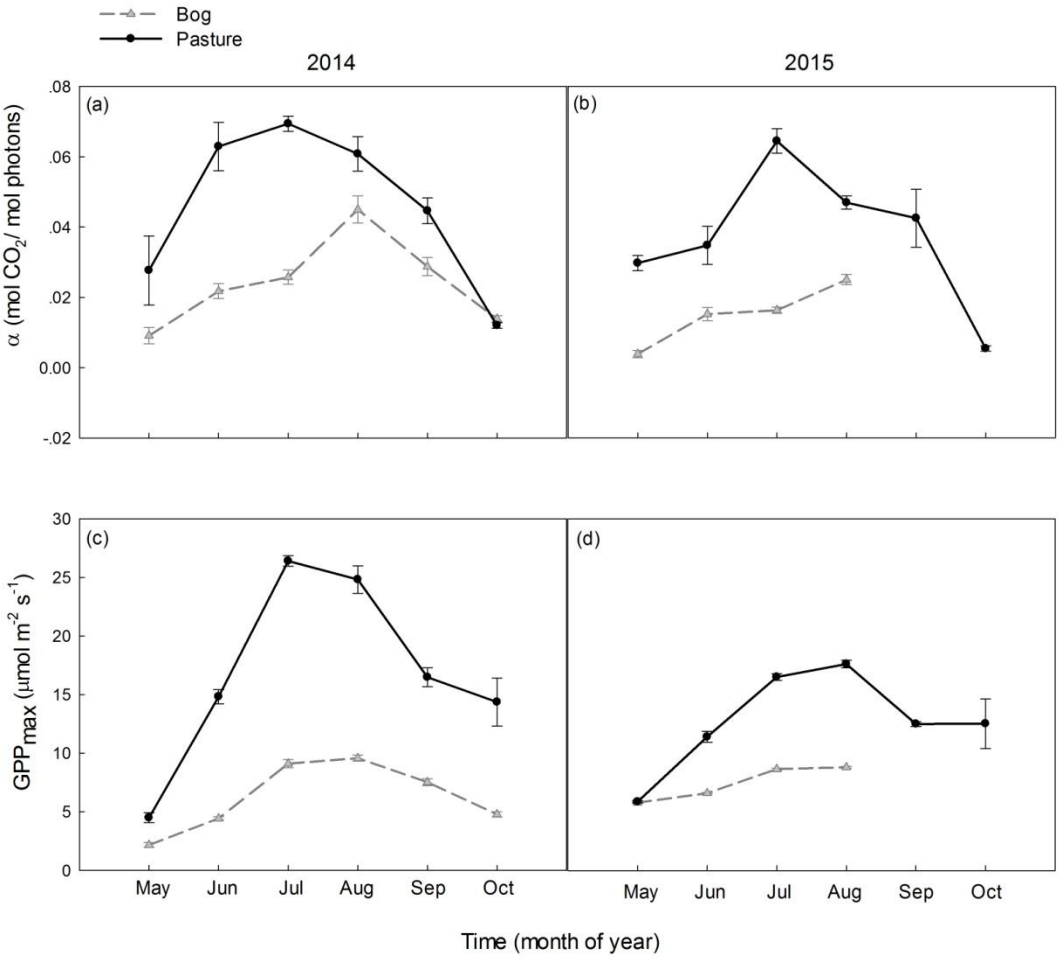


Fig.5.7 Monthly light use efficiency (α) and maximum carbon fixation rate (GPP_{max}) for the bog and abandoned peatland pasture.

5.4.3. Controls on the between-site difference in CO_2 flux

During the growing season, the measured nighttime ER decreased linearly with SWC_{30}, but increased exponentially with T_1 for both sites (Fig.5.8: a, b, e, f). ER at the bog and pasture had a stronger correlation with T_1 (R^2: 0.59 for the bog and 0.84 for the pasture) than SWC_{30} (R^2: 0.11 for the bog and 0.23 for the pasture) (Fig.5.8: a, b, e, f) and ER at the pasture was more sensitive to soil temperature change (Q_{10} = 3.3), compared with that of bog (Q_{10} = 2.2) (Fig.5.8: b, f). Daytime average GPP was more strongly related to T_a than to PPFD at both sites, with T_a

explaining 57% of the variation in GPP at the bog and 63% at the pasture, while PPFD only accounted for ~30% of the explained variance at both sites (Fig.5.8: c, d, g, h). Moreover, examining differences in fluxes between sites, the difference in the daily average GPP between the two sites (ΔGPP) explained 93% and 92% of the between-site difference in the daily average NEE (ΔNEE) and ER (ΔER) (Fig.5.9: a, b). The between site difference in NEE (ΔNEE) was highly related to ΔGPP (R^2 = 0.93), but only moderately related to ΔER (R^2 = 0.59) (Fig.5.9: b, c).

Although no significant inter-annual difference in the end of the growing season was found for the pasture, the start of the growing season in 2015 was 15 days later than in 2014, causing a shorter GSL in 2015 than in 2014 (Fig. S5.1, Table 5.2). GPP peaked near DOY (day of year) 207 in 2014 and DOY 212 in 2015 at the bog and at the pasture GPP peaked near DOY 205 in 2014 and DOY 207 in 2015. The growing season of the bog began earlier and ended later than that at the pasture, resulting in a longer growing season length of 223 days in 2014 compared with only 193 days in 2014 and 177 days in 2015 at the pasture (Fig.S5.1, Table 5.2), suggesting that the growing length was not the cause of the difference in NEE between the two sites.

The 16-day average EVI revealed similar seasonal patterns for the two sites. For both sites EVI began to increase at DOY 105 in 2014 and DOY 121 in 2015, peaking at DOY 217 and then decreasing into the autumn in both years. The time when EVI began to increase coincided with the start of growing season at the bog in 2014 (DOY 101), but about 20 days earlier than that for the abandoned peatland pasture in 2015 (DOY 129) (Table 5.2, Fig.5.10). The EVI peak time was in line with the GPP peak date in both years at both sites. The date when EVI started to

decrease was consistent with the end of growing season date at the bog, but about 20 days later than that at the abandoned peatland pasture. Moreover, the EVI of the abandoned peatland pasture was higher than that in the bog during growing season period, especially in 2014. The difference in EVI (ΔEVI) between the two sites explained 46%, 45% and 63% of the explained variation in ΔNEE, ΔER and ΔGPP, respectively (Fig.5.10). By comparison, the difference in WT (ΔWT) explained 52% of the variation in ER (ΔER) between the two sites (Fig.5.11).

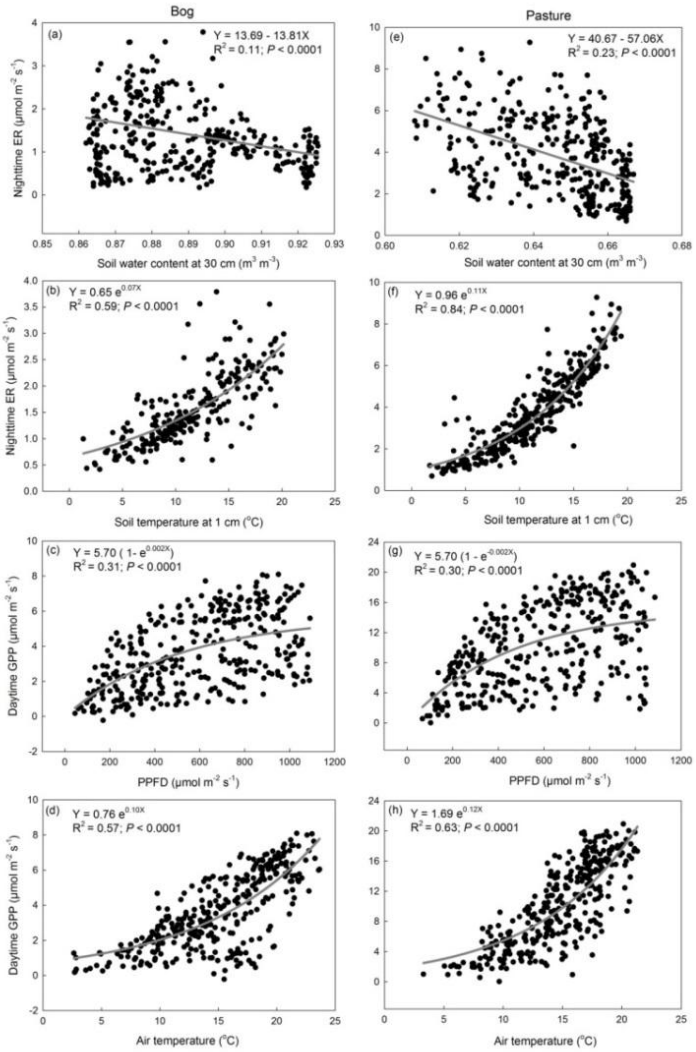


Fig.5.8 The dependency of growing season nighttime ER and daytime GPP on environmental variables at the bog (a-d) and abandoned peatland pasture (e-h).

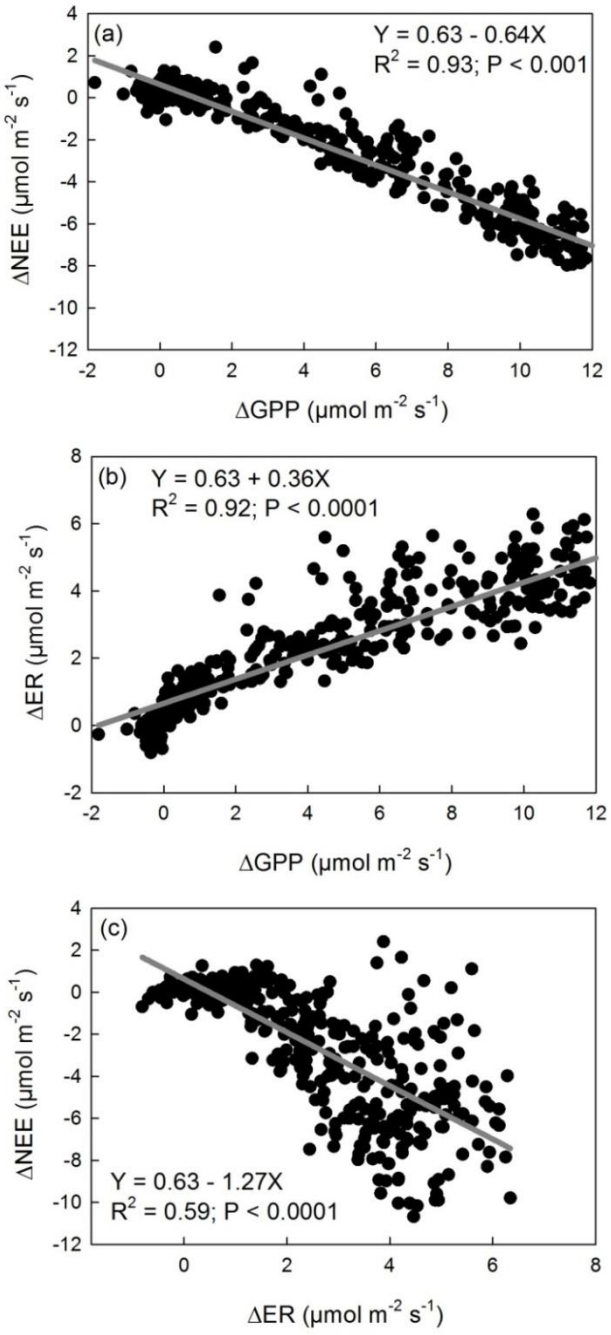


Fig.5.9. The relationship between pasture minus bog differences in NEE (ΔNEE) and differences in the flux components ΔER and ΔGPP.

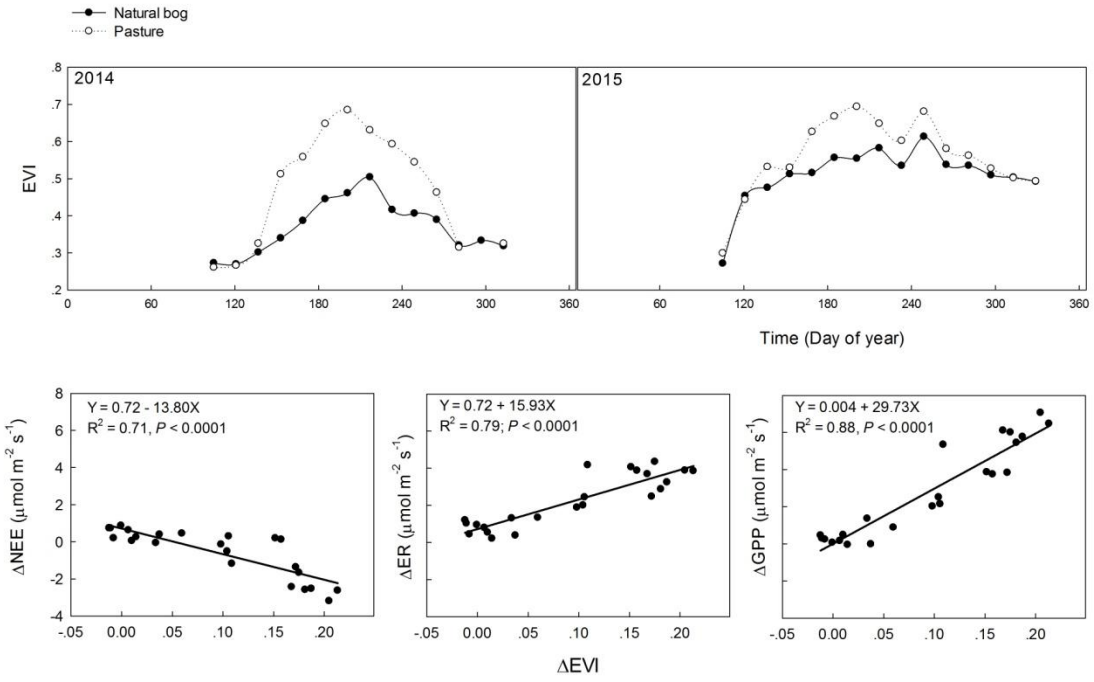


Fig.5.10 Comparison of enhanced vegetation index (EVI) between the bog and the abandoned pasture from 2014 to 2016 (top panels) and the relationship between EVI difference (pasture minus bog, ΔEVI) and ΔNEE, ΔER and ΔGPP.

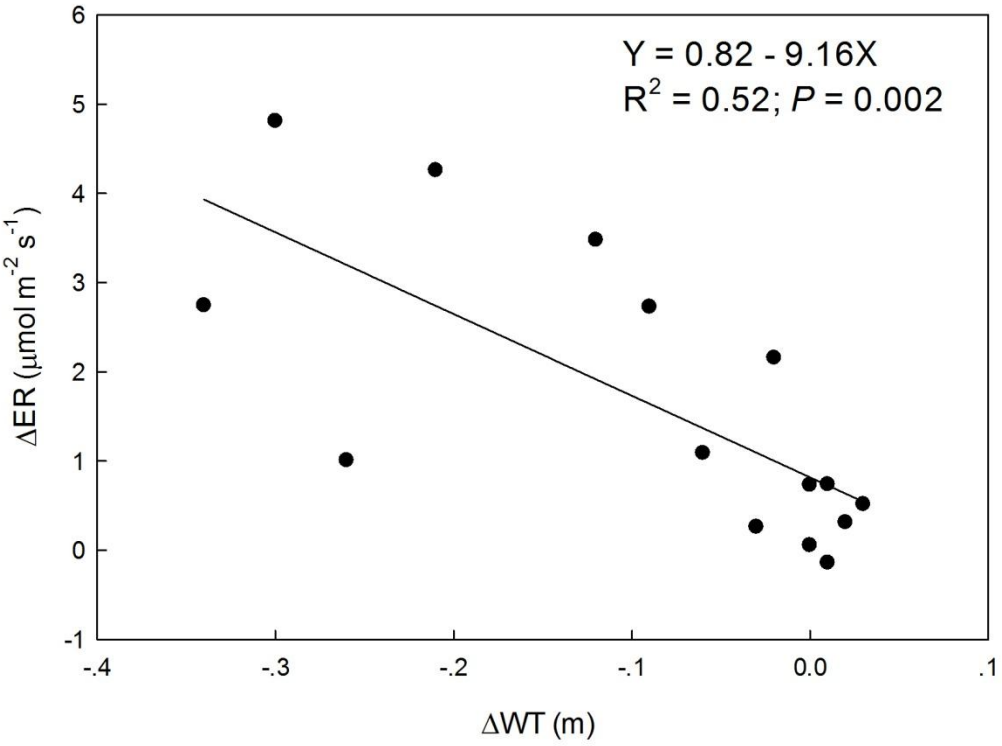


Fig.5.11 The relationship between the difference in monthly average water table depth (ΔWT) and the difference in ER (ΔER) between the bog and abandoned pasture.

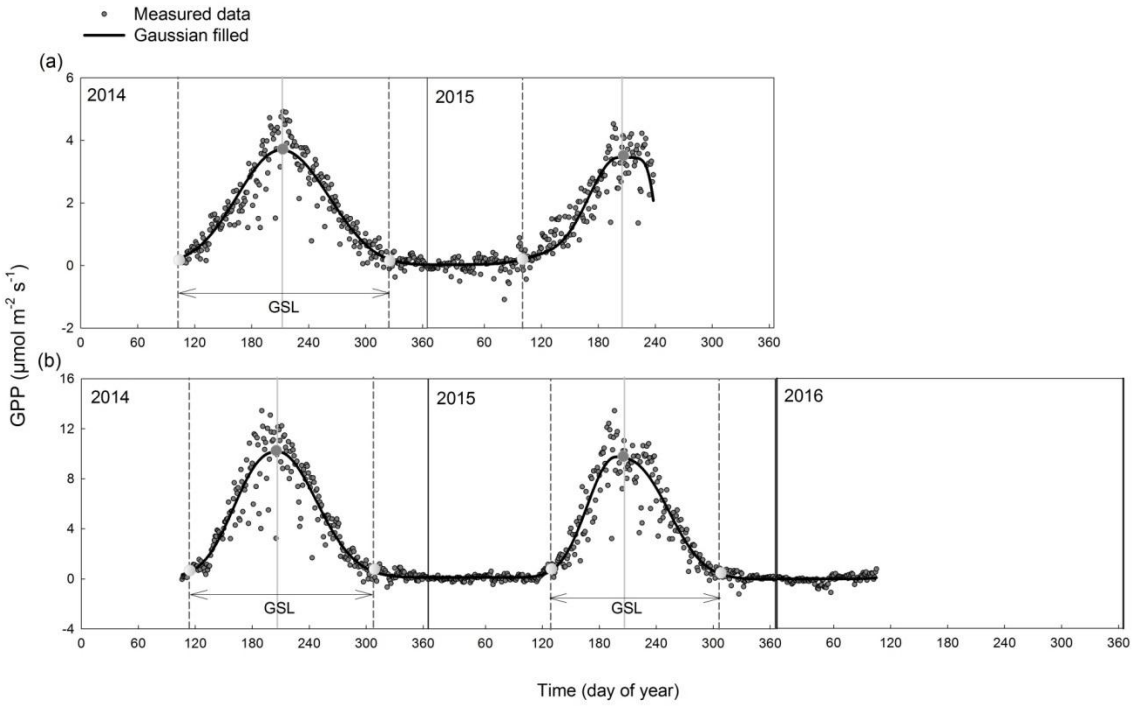


Fig.S5.1 TIMESAT phenological indices extracted from the time series of GPP at (a) the bog and (b) abandoned peatland pasture. The dashed grey line indicated the beginning and end of the growing season, the solid grey line indicates the date when the maximum GPP occurred. GSL is the growing season length.

Table 5.2 The phenological parameters derived for the bog and abandoned peatland pasture based on gross primary productivity (GPP) using TIMESAT software. GSS, GSE, GSL indicate the start of growing season, the end of growing season and the growing season length, respectively. Peak rate is the highest GPP value and peak time is the day when peak rate occurs

Site	Year	GSS	GSE	GSL	Peak rate	Peak time
		Day of year		Days	μmol m^{-2} s^{-1}	Day of year
Bog	2014	101	324	223	3.7	212
	2015	100			3.45	207
Pasture	2014	114	307	193	10.17	205
	2015	129	306	177	9.75	207

5.4.4. Comparison of the annual budget of NEE, ER and GPP between the two sites

GPP and ER of the abandoned peatland pasture were significantly higher than that at the bog during the growing season, but no significant difference was found in the non-growing season when the cumulative NEE and GPP of both sites were not significantly different from zero and the cumulative ER was very low but significantly positive (Table 5.3). On an annual basis, the abandoned peatland pasture acted as a CO_2 sink of -85 ± 14 g C m^{-2} yr^{-1} in 2014-15 and -82 ± 37 g C m^{-2} yr^{-1} in 2015-16, significantly higher than that of -18 ± 14 g C m^{-2} yr^{-1} in 2014-15 at the bog. Similarly, both annual GPP (1106 ± 144 g C m^{-2} yr^{-1} in 2014-15 and 990 ± 124 g C m^{-2} yr^{-1} in 2015-16) and ER (1021 ± 138 g C m^{-2} yr^{-1} in 2014-15 and 908 ± 118 g C m^{-2} yr^{-1} in 2015-16) were about 2.5 times the annual fluxes at the bog (Table 5.3).

Table 5.3. The total accumulated NEE, ER and GPP (g CO_2-C m^{-2}) for the growing season, non-growing period and the annual budget in two years from April 2014 to April 2015 for the bog and from April 2014 to April 2016 for the abandoned peatland pasture.

Site	Period	2014-15			2015-16		
		NEE	ER	GPP	NEE	ER	GPP
Bog	Growing season	-99 ±57	342 ±51	441 ±76	n/a	n/a	n/a
	Non-growing season	80 ±93	87 ±37	8 ±100	n/a	n/a	n/a
	Whole year	-18 ±14	431 ±84	449 ±85	n/a	n/a	n/a
Abandoned peatland pasture	Growing season	-215 ±94	863 ±90	1080 ±131	-182 ±70	787 ±92	977 ±116
	Non-growing season	130 ±117	159 ±77	28 ±140	108 ±122	121 ±58	13 ±135
	Whole year	-85 ±40	1021 ±138	1106 ±144	-82 ±37	908 ±118	990 ±124

5.5. Discussion

5.5.1. Comparison of the controls over the temporal variability of CO_2 flux at the two sites

We did not find significant differences in the environmental controls on the temporal variability of the daily average CO_2 fluxes at the bog and pasture, though the daily average ER during the growing season at the pasture was more sensitive to changes in T_1 than at the bog. The daytime GPP increased with the increase in T_a and PPFD at both sites, which has been well documented in previous researches (Flanagan and Syed, 2011; Schubert et al., 2010; Teh et al., 2011). The dependency of daytime GPP on PPFD was weaker than T_a at both sites, similar to previous studies (Lund et al., 2012; Petrone, et al., 2003). The weak correlation between PPFD and daytime GPP was due in part because PPFD was highly variable, with the PPFD ranging from near zero to >700 $\mu mol\ m^{-2}\ s^{-1}$ throughout the growing season (Fig.5.2). In addition, the weaker correlation between PPFD and daytime GPP is likely due to the mismatch in timing of peak PPFD and peak GPP. PPFD peaked near mid-late June (Fig.5.2) when plants were not fully developed and had limited CO_2 uptake capacity and we found the highest GPP and EVI values occurred in late July for both sites. In order to exclude the phenology effect, we examined the function of PPFD on GPP in the early (May and June), middle (July and August) and late growing season (September and October) and found that PPFD was a strong control over GPP only during the middle growing season, when it explained 55% and 76% of GPP variation at the bog and pasture, respectively (Fig.S5.2).

Nighttime average ER was strongly related to T_1 and the dependency of ER on SWC_{30} was weak at both sites (Figs. 5.9, 5.10). Similarly, the reduced dependency of ER on soil moisture (water table/SWC) has been found in other bogs (Lafleur et al., 2005a). We found that ER of the pasture

was more sensitive (Q_{10} = 3.32) to soil temperature change when compared with that of the bog (Q_{10} = 2.23). Previous studies have shown that the temperature sensitivity of labile C is greater than that of recalcitrant C (Luo et al., 2001; Melillo et al., 2002). Specific UV absorbance at 254 nm ($SUVA_{254}$) is an index used to assess DOC composition, with low/high $SUVA_{254}$ indicating low/high aromaticity and the lower the $SUVA_{254}$ of DOC, the easier it is to decompose (Weishaar et al., 2003). We found that the $SUVA_{254}$ of DOC in the pasture was significantly lower than that in the bog (Fig. S5.3b), suggesting that the pasture had more labile C than the bog, which may be in part a result of the lower average water table at the pasture as suggested by Kwon et al. (2013). Therefore, the higher sensitivity of ER to temperature change in the pasture possibly resulted from a greater availability of labile C.

During the non-growing season, GPP and ER at both sites showed weak dependency on the environmental variables (Fig.S5.4). Similarly, the lack of strong dependency of ER on the environmental variables in the non-growing season was observed in many previous field studies, probably caused by the very narrow range in environmental conditions (Alm et al, 1999; Merbold et al., 2012; Schindlbacher et al., 2014). At our sites, non-growing season SWC was high and almost constant and T_1 was low, <5 ^{o}C for both sites (Fig. S5.4). In addition, we found high CO_2 emissions occurred when T_1 was around 0 ^{o}C for both sites (Fig. S5.4:e, f), which may be another mechanism weakening the relationship between CO_2 flux and environmental variables. The large burst of CO_2 with ground temperatures of around 0^{o}C during soil thawing period was previously found in a wide range of ecosystems including peatlands (Bubier et al., 2002; Lund et al., 2005), forest (Skogland et al., 1988), agricultural systems (Lange et al., 2016; Priemé and Christensen, 2001), tundra (Mikan et al., 2002; Raz-Yaseef et al., 2017) and alpine

meadow (Kato et al., 2005), due to either stimulation of microbial decomposition associated with freeze–thaw dynamics (Lange et al., 2016; Lund et al., 2005) or release of trapped/ stored gas (Friborg et al. 1997; Mast et al., 1998; Raz-Yaseef et al., 2017) or both (Bubier et al. 2002). However, the underlying mechanisms for this phenomenon of flux burst during soil thawing is not well understood and warrants more study at our sites.

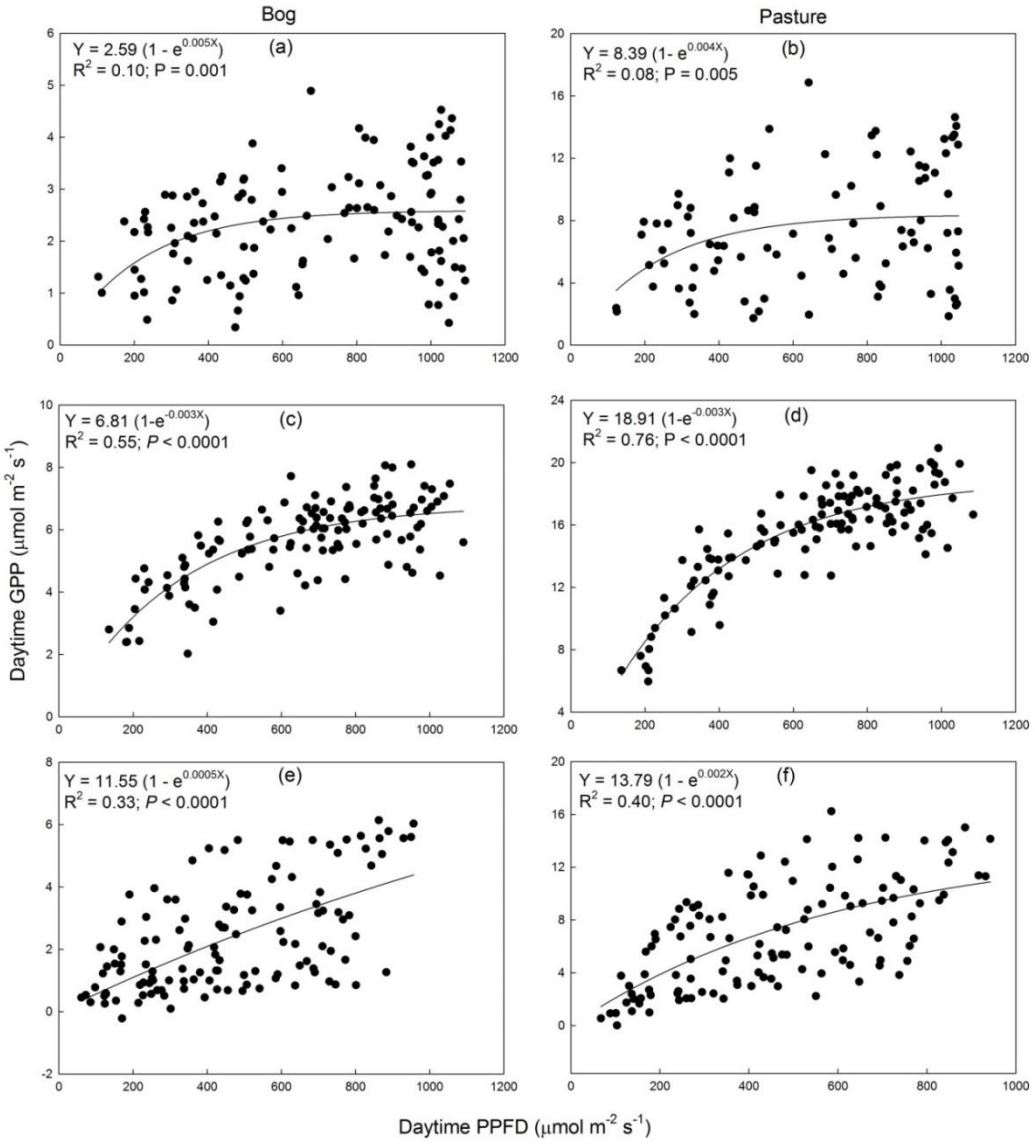


Fig. 5.S2 The relationship between daytime GPP and PPFD in the early (a, b), middle (c, d) and late (e, f) growing season at the bog and pasture

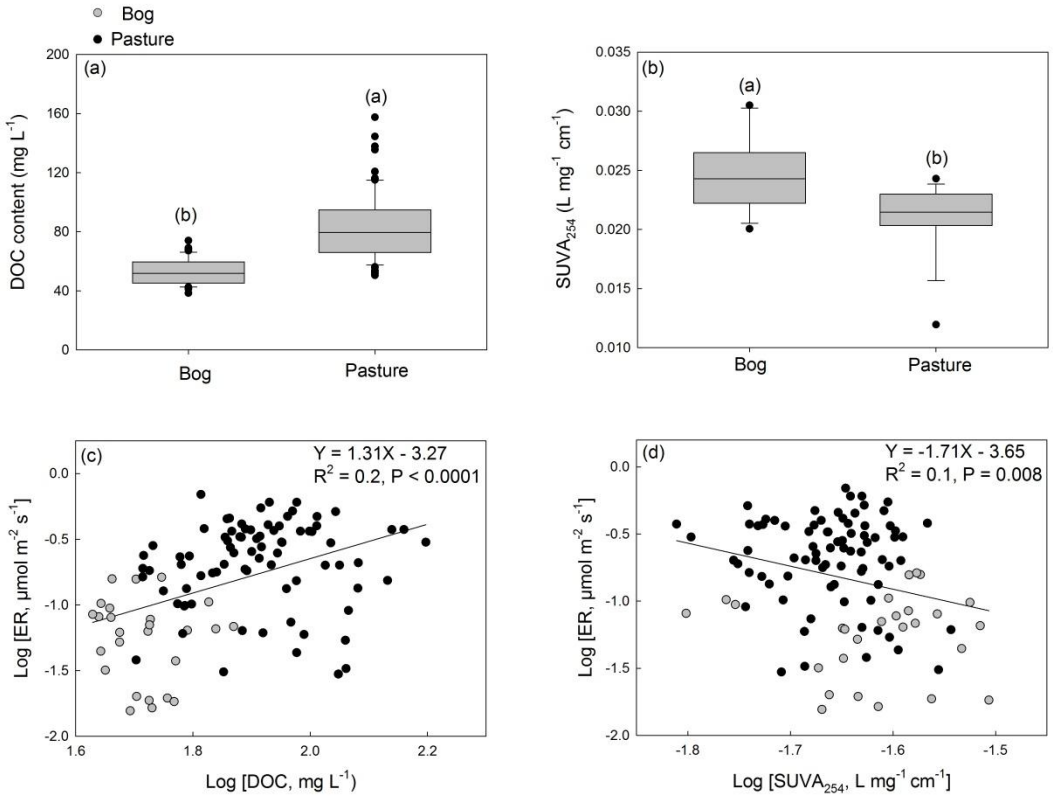


Fig. 5.S3 The comparison of dissolved organic carbon (DOC) content (a) and $SUVA_{254}$ (b) between the bog and abandoned peatland pasture; the relationship between ER (c) and DOC and $SUVA_{254}$ (d).

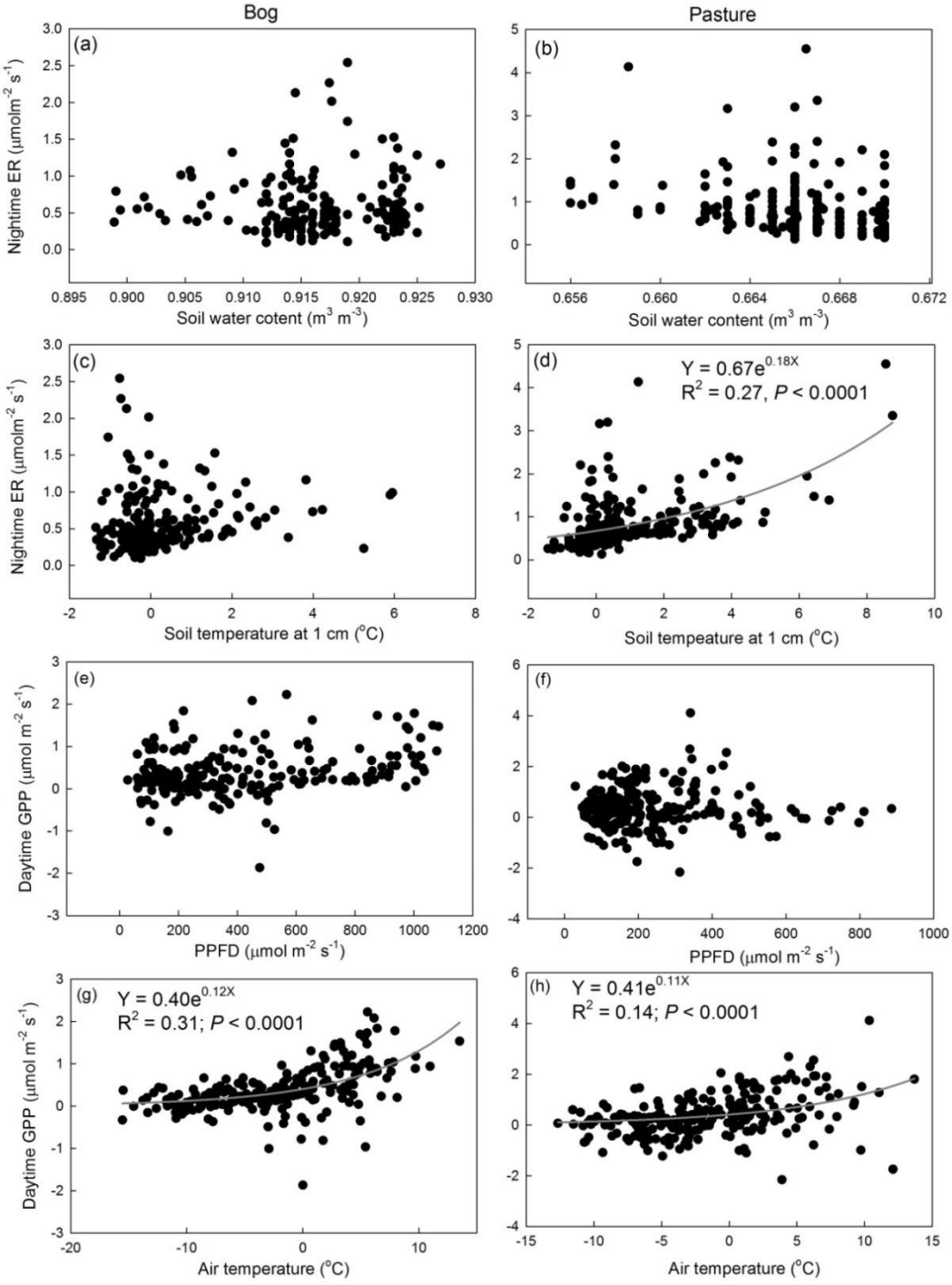


Fig.5.S4 The dependency of non-growing season nighttime ER and daytime GPP on environmental variables at the bog (a, c, e, g) and abandoned peatland pasture (b, d, f, h).

5.5.2. Comparison of NEE, GPP and ER between the bog and pasture

Although both GPP and ER at the pasture were substantially higher than those at the bog, the between-site GPP difference was greater, making the pasture a stronger CO_2 sink (82-85 g C m^{-2} yr^{-1}) than the bog (18 g C m^{-2} yr^{-1}). The annual GPP at the pasture (1106 ± 144 g C m^{-2} yr^{-1} in 2014-15; 990 ± 124 g C m^{-2} yr^{-1} in 2015-16) was ~2.5 times of that at the bog (449 ± 85 g C m^{-2} yr^{-1} in 2014-15). We attribute this difference to different vegetation conditions, where the pasture had two times higher aboveground biomass than that at the bog (Luan and Wu, 2015). EVI, as a measure of vegetation cover and biomass, reflects this difference as well. Hence, the variation in ΔEVI between the two sites was closely related to the variation in ΔGPP (Fig.5.9). The annual ER budget at the pasture (1021 ± 138 g C m^{-2} yr^{-1} in 2014-15; 908 ± 118 g C m^{-2} yr^{-1} in 2015-16) was significantly higher than that at the bog (431 ± 84 g C m^{-2} yr^{-1} in 2014-15). We found that 93% of the variation in monthly average ΔER between the two sites can be explained by the ΔGPP (Fig.5.9b), suggesting that the difference in ER was partly due to the difference in autotrophic respiration for the growth and maintenance of plants between the two sites. This hypothesis was supported by previous findings that plant respiration was positively related to its photosynthesis (Amthor and Baldocchi, 2001; Bubier et al., 1998; Flanagan and Johnson, 2005). In addition, ER at the pasture was enhanced because of its lower WT, with the between-site ΔWT explaining 52% of the ΔER variation (Fig.5.11). The lower water table promotes peatland ER by exposing more peat material to aerobic conditions and stimulates their mineralization (Chivers et al., 2009; Salm et al., 2012). Finally, when compared to the bog, the higher ER at the pasture was also likely due to a greater quantity of substrates, which provided more sources for CO_2 release (Murphy and Moore, 2010). This hypothesis is supported by our data that the DOC content in the pasture was significantly higher than that in the bog (Fig.S5.3a), similar to previous studies (Holden et al.,

2004; Yang et al., 2017).

5.5.3. Comparison of the annual CO_2 budget between our two sites and other peatlands

The NEE of this bog (-18 ± 14 g C m^{-2} yr^{-1} in 2014-15) was within the lower range of previously reported CO_2 uptake for undisturbed temperate and boreal peatlands (Table 5.4). Most agriculturally managed peatlands have been found to be "hotpots" of CO_2 emissions (Aslan-Sungur et al., 2016; Gronlund et al., 2008; Hargreaves et al., 2003; Knox et al., 2015; Lohila et al., 2004; Maljanenet et al., 2001), though exceptions exist, such as an agricultural peat meadow in the Netherlands (Hendricks et al. 2007). Moreover, a few previous studies have compared the CO_2 flux between undisturbed and managed (drained or rewetted) peatland sites and found that drainage induces CO_2 loss from peatlands (Strachan et al., 2016; Wilson et al., 2016). However, at our site the abandoned pasture acted as a stronger CO_2 sink than the adjacent bog (NEE: -82 ~ -85 g C m^{-2} yr^{-1} at the pasture and -18 g C m^{-2} yr^{-1} at the bog). The difference may be due in part because previous reports of CO_2 fluxes were based on drained peatlands with active usage such as mowing, grazing, tillage and manuring, and these management interventions can promote C loss from drained peatlands (Knox et al., 2015; Lohila et al., 2004; Nieveen et al., 2005). In addition, the high CO_2 uptake capacity of this pasture can also be attributed to the high primary productivity of the dominant plant RCG, since its primary productivity was found to be higher than values derived from the global grasslands (Scurlock et al., 1999; Shurpali et al., 2010). It has been suggested that cultivation with a perennial RCG bioenergy crop on drained peatland soils as an after-use option is highly beneficial for increasing CO_2 sequestration (Järveoja et al., 2015; Karki et al., 2016; Shurpali et al., 2009, 2010). For example, the average CO_2 sink capacity of a cultivated peat soil ecosystem in eastern Finland was ~100 g C m^{-2} yr^{-1} in four study years of 2004-2007 (Shurpali et al., 2009), similar to that of 82-85 g C m^{-2} yr^{-1} for our

pasture.

5.6. Conclusion

Although several studies of CO_2 flux have been performed on agriculturally managed peatlands, our study, to our best knowledge, this study may be the first to evaluate the impact of agricultural management and abandonment on CO_2 sequestration capacity of boreal peatlands.

Our results indicate that the abandoned peatland pasture had much greater CO_2 sink strength than the undisturbed bog and can sequestrate CO_2 as efficiently as some restored peatlands. This finding raises the intriguing possibility that abandonment of peatland pasture could be an effective management decision for increasing landscape C sequestration. However, several aspects of C exchange not studied here should be considered. First, inter-annual variation in CO_2 flux can be large and with only two years of measurements the limits of CO_2 sequestration and the influence of extreme climate variation on CO_2 exchange in this ecosystem are unknown. Second, CO_2 flux is only one component of the total net ecosystem carbon budget (NECB) and greenhouse gases (GHGs), which ultimately will determine the pasture's influence on the atmosphere radiation balance. The exchange of other important greenhouse gases, such as methane and nitrous oxide as well as the lateral input/export of water-borne DOC, will determine the overall NECB and GHGs. At present, little is known about these fluxes in abandoned peatland pastures and future studies are needed to assess their importance.

Chapter 6 - Can an abandoned peatland pasture be a stronger greenhouse gas sink than an adjacent undisturbed boreal bog in western Newfoundland, Canada?

6.1. Abstract

Agriculturally managed peatlands have been identified as "hotspots" for greenhouse gas emissions and great efforts have been made to mitigate their emissions. Here we measured landscape-scale carbon dioxide (CO_2) and methane (CH_4) fluxes by eddy covariance and plot-scale nitrous oxide (N_2O) fluxes using static chamber technique in a boreal bog and an adjacent abandoned peatland pasture to determine the effect of agricultural conversion and abandonment on the carbon and greenhouse gases (GHGs) flux of a boreal bog. The pasture was a stronger CO_2 sink and smaller CH_4 source, accumulating a total of 167 g CO_2-C m^{-2} and emitting a total of 0.37 g CH_4-C m^{-2} during the two study years, when compared to the CO_2 uptake of 17 g C m^{-2} and CH_4 emission of 6.7 g CH_4-C m^{-2} for the bog. Both the bog and pasture had very low growing season N_2O fluxes, accumulating -0.050 ± 0.036 g N_2O -N in 2014 and -0.066 ± 0.193 g N_2O -N in 2015 at the bog and -0.004 ± 0.045 g N_2O -N in 2014 and -0.080 ± 0.199 g N_2O -N in 2015 at the pasture. The carbon and GHGs balance were mainly determined by the magnitude and direction of CO_2 for the pasture, but the GHGs balance was determined by CH_4 flux for the bog. The pasture acted as a stronger carbon and GHGs sink, with the carbon balance of -167 g C m^{-2} and (sustained-flux) global warming potential (GWP/SGWP) of -174/-182 g C m^{-2} g CO_2-C equivalents m^{-2} during the two study years, when compared to the bog with the accumulative carbon of -10 g C m^{-2} and GWP/SGWP of 131/174 g CO_2-C equivalents m^{-2}, respectively.

Therefore, our results suggest that the carbon accumulation capacity and "climate cooling" function of agriculturally managed peatlands can become stronger than the undisturbed peatlands after long-term abandonment.

6.2. Introduction

Agricultural management involving drainage, cultivation and fertilization has widely affected the northern peatlands (Oleszczuk et al., 2008), with approximately 14-20% (nearly 3×10^5 km^2) of these ecosystems being converted for meadows and pastures (Lappalainen, 1996; Worrall et al., 2010). The worldwide drainage of peatlands for agriculture has resulted in significant peat soil subsidence due to either changes in physical conditions or enhanced oxidation rate of peats (Couwenberg, 2011; Couwenberg and Hooijer, 2013; Dawson et al., 2010; Leifeld et al., 2011; Pronger et al., 2014), making agriculturally managed peatlands 'hotspots' for greenhouse gases (GHGs) emissions and motivating great efforts to study their role in regional and global carbon budgets.

Carbon dioxide (CO_2) is usually the most important GHG in the peatland carbon budget and the net ecosystem exchange of CO_2 (NEE) is determined by two main processes: gross primary productivity (GPP) where ecosystems incorporate CO_2 into biomass via plant photosynthesis and ecosystem respiration (ER) which is the loss of CO_2 from ecosystems via autotrophic and heterotrophic respiration to the atmosphere. The effects of drainage on these processes are complex. On one hand, drainage exposes more peats to aerobic conditions and is expected to promote peat mineralization and increase C loss as CO_2 (Ballantyne et al., 2014; Frolking et al., 2011; Joosten, 2009; Salm et al., 2012). On the other hand, plant cultivation on drained peatlands

or plant regeneration of abandoned agriculturally managed peatlands enhances photosynthesis and results in C sequestration by the ecosystem (Hendriks et al., 2007; Shurpali et al., 2009). The effect of agriculture management on the CO_2 flux of peatlands depends on the net response of the two CO_2 flux processes. Although many studies have shown that agricultural management turns peatlands from net CO_2 sinks to sources (Campbell et al., 2015; Grønlund et al., 2008; Leiber-Sauheitl et al., 2014; Teh et al., 2011), exceptions exist where a few studies have reported that enhanced gross primary productivity outweighed elevated ER following agricultural management (Hendriks et al., 2007; Karki et al., 2014).

Methane (CH_4) is the second most prominent GHG, with its global warming potential (GWP) 25 times that of CO_2 on 100 year time horizon (Forster et al., 2007). CH_4 fluxes of peatlands are determined by the processes of CH_4 production, consumption and transport (Lai, 2009). The production of CH_4 by methanogens requires anaerobic conditions (Dalal and Allen, 2008), however CH_4 oxidation occurs in aerobic conditions. The relative importance of these two processes is closely linked to water table depth in peatlands (Lai, 2009). Drainage, which lowers the water table, causes low CH_4 production and high CH_4 oxidation potential, usually resulting in low CH_4 emission (Lai, 2009). Moreover, agriculturally managed peatlands have high concentrations of vascular plants as a result of cultivation or vegetation succession and the drier conditions favor the transport of CH_4 via plant arenchyma (Joabsson et al., 1999; Ström et al., 2005). On balance, however, many studies suggest CH_4 emissions are inhibited by agriculture management and drainage causes peatlands to be a small CH_4 source or even a small sink of CH_4 (Bussell et al., 2010; Knox et al., 2015; Maljanen et al., 2010; Schrier-Uijl et al., 2014; Smith and Conen, 2004).

Nitrous oxide (N_2O) is another important greenhouse gas, with the GWP 298 times that of CO_2 on 100 year time basis (Forster et al., 2007). N_2O is produced in soils mainly by microbial activities, nitrification and denitrification being the key processes (Priemé and Christensen, 2001). Pristine peatlands emit small amounts of N_2O because they have low background soil N content and they can even be net sinks for this gas (Martikainen et al., 1993, Regina et al., 1996). However, N_2O plays an important role in nutrient-rich managed peatlands. On one hand, the water table drawdown of managed peatland increases the availability of oxygen and mineral nitrogen, which favors N_2O production (Martikainen et al., 1993). On the other hand, nitrogen application also promotes the production of N_2O due to increased nutrient substrate availability (Alm et al., 2007, Van Beek et al., 2011). Agriculturally managed peatlands have been found to be large N_2O sources (Alm et al., 2007, Salm et al., 2012, Teh et al., 2011) and N_2O exchange cannot be neglected when estimating their GHGs budgets.

During past decades, progress has made on the effects of agricultural management on the GHGs flux of peatlands (Haddaway et al., 2014; Maljanen et al., 2010; Salm et al., 2012; Teh et al., 2011). Agriculture management is proposed to increase the emissions of GHGs, mainly by promoting the emissions of CO_2 and N_2O. However, the increase magnitude of GHGs emissions following agriculture management was found to be significantly variable, dependent on the management intensity, peatland initial conditions, cultivation species, time for plant regeneration and fertilization amount (Beetz et al., 2013; Maljanen et al., 2010; Schrier-Uijl et al., 2014). Moreover, the knowledge of how agricultural management affects GHGs fluxes is limited by insufficient direct comparisons of GHGs between undisturbed peatlands and agriculturally

managed ones and failure to consider all three GHGs species (Haddaway et al., 2014).In addition, most studies have focused on Nordic countries (Maljanen et al., 2010), the USA (Knox et al., 2015; Teh et al., 2011) and the Netherland (Schrier-Uijl et al., 2014), however, such knowledge is limited in other regions of the globe. The above mentioned limitations contribute to uncertainty in investigating the effect of agricultural management on the GHGs fluxes of peatlands at regional and global scales. Therefore, more comparative studies of GHGs fluxes between agriculturally managed peatlands and undisturbed ones are urgently needed.

In Canada, about 170,000 km^2 of peatlands have been converted for agricultural management as vegetable and small fruit production and pasture, accounting for 15% of the total national resource of peatlands and mires (Joosten, 2009; Oleszczuk et al., 2008). The amount of C loss in the form of CO_2 from Canadian peatlands drained for agriculture was estimated at 4.9 Mt CO_2 yr^{-1} (Joosten, 2009). However, this estimation was based on limited subsidence measurements (Couwenberg, 2009) and is highly uncertain. To the best of our knowledge, few studies have reported landscape-scale CO_2 and CH_4 fluxes or full GHGs fluxes from agriculturally managed peatlands in Canada, nor are there any direct comparison of GHGs fluxes between undisturbed peatlands and managed ones.

This study aims to fill the knowledge gap on the impacts of agriculture management and abandonment on peatland GHG emissions by providing full GHGs balance between a boreal bog and an adjacent abandoned peatland pasture, which was a part of the same peatland several decades earlier. Our assessment is based on two years of data from April 2014 to April 2016 and consists of landscape-scale CO_2 and CH_4 fluxes via eddy covariance (EC) measurements and

N_2O flux during the growing season of 2013-2016 from static chamber measurements. We hypothesized that previous agricultural management and vegetation succession at the abandoned peatland pasture would favor the emissions of CO_2 and N_2O, but would lower CH_4 emission compared to the undisturbed bog; and the abandoned peatland pasture would be a smaller C and GHGs source than other intensively managed peatlands since plant regeneration on managed peatalnds after abandonment can reduce the emission of CO_2 and N_2O flux (Hendriks et al., 2007; Schrier-Uijl et al., 2014).

6.3. Methods

6.3.1. Study site

Our site is located in the Robinsons pasture, western Newfoundland, Canada (48.264 N, 58.665 W). According to the data from the nearest weather station at Stephenville (48.541 N, 58.55 W) (http://climate.weather.gc.ca/climate_normals/results_1981_2010_e.html?stnID=6740&autofwd=1), the average annual air temperature and precipitation are 5 °C and 1340 mm for the past 30 years [1981 to 2010], respectively. The mean annual precipitation was partitioned into 995 mm as rainfall and 393 mm as snowfall. The growing season (from May to October) temperature and precipitation averaged about 12 °C and ~705 mm. Our research sites are in a peatland complex and comprise an undisturbed bog and an abandoned peatland pasture (Fig.6.1). The pasture (~0.2 km^2) was originally an ombrotrophic bog that was drained in the 1970s by a network of ditches (~0.5 m in depth and ~30 cm in width) along an east-west transect with the distance of 20-30 m between ditches. At the time of drainage pasture forage grasses were introduced. The site was used as pasture for 10 years and then abandoned. After abandonment, the site was left to regenerate for ~25 years, but with active drainage. The undisturbed bog (0.36 km^2) was located

in the adjacent east of the abandoned peatland pasture. In October 2013, we installed the eddy covariance systems at the bog and pasture to study how agriculture management including drainage and vegetation composition change affects the landscape CO_2 and CH_4 fluxes of boreal peatland ecosystems in Canada, where the undisturbed bog acted as a control. A detailed description of vegetation and site characteristics was provided by Luan and Wu (2014) and the following is a brief summary of that information. The undisturbed bog belongs to the typical peatland type in Newfoundland, with the component landforms of hollows, hummocks and pools and a substrate dominated by brown bog mosses *(Sphagnum warnstorfii and Sphagnum capillifolium)* and partly with gray reindeer lichens *(Cladina spp.)*. A high abundance of ericaceous shrubs and herbs/sedges grow in both hollows and hummocks, but hollows are dominated by sedges and hummocks by shrubs. The dominant species of herbs and sedges are *Muhlenbergia rigens, Rubus chamaemorus, Lycopodiopsida* and *Trichophorum cespitosum*, and the main ericaceous shrubs include *Gaylussacia spp., Empetrum nigrum, Kalmia angustifolia, Rhododendron groenlandicum, Cornus canadensis and Andromeda polifolia*. The abandoned peatland pasture is a mosaic of patches, which are dominated by reed canary grass (RCG: *Phalaris arundinacea*), various low herbaceous and graminoid species (*Carex* spp., *Ranunculus acris, Ranunculus repens, Hieracium* spp.) and several dwarf shrubs and tall shrubs [sweet gale (*Myrica gale*), labrador tea (*Rhododendron groenlandicum*), mountain fly honeysuckle (*Lonicera villosa*), rhodora (*Rhododendron canadense*), and chokeberry (*Photinia* spp.)]. In 2013, the dry aboveground biomass in the bog was estimated at 197 ± 87 g m^{-2} for hummocks and 191 ± 41 g m^{-2} for hollows, both of which were significantly lower than that of 591 ± 246 g m^{-2} for patches dominated by dwarf shrubs and tall grass in the pasture (Luan & Wu, 2015).

6.3.2. CO_2 and CH_4 flux and meteorological measurements

Identical EC systems were operated at the bog and pasture for the period from April 2014 to April 2016 (Fig.6.1). Each system consisted of a three-dimensional (3-D) sonic anemometer (Gill WindMaster, Gill Instruments Ltd, Lymington, Hampshire, UK), a fast response infra-red gas analyzer (IRGA: LI-7200 Enclosed CO_2/H_2O Analyzer, Li-Cor Inc., Nebraska, USA) and an open path infrared gas analyzer (LI-7700, Li-Cor Inc., Nebraska, USA). The 3-D sonic anemometers measured wind speed (u, v, w) and direction and sonic temperature at 3.44 m height for the bog and 3.7 m height for the abandoned peatland pasture. The LI-7200 analyzers were mounted at the height of 3.21 m at the bog and 3.54 m at the pasture to simultaneously measure variations in CO_2 and H_2O molar densities. The LI-7700 open path CH_4 sensor based on wavelength modulation spectroscopy was mounted at 3.34 m height in the bog and 3.6 m height in the pasture to measure CH_4 concentration. The separation between the sonic anemometer and LI-7200 was 3.5 cm northward, 3.5 cm eastward and 23 cm vertically for the bog and 7.07 cm northward, 7.07 cm eastward and 16 cm vertically for pasture. The separation of LI-7700 from sonic anemometer was 17cm northward, -1cm eastward and 10cm vertically, respectively, for the bog but the northward separation was 18 cm for the pasture. For the enclosed LI-7200, air was pulled by a diaphragm pump through a 1 m long sample tube to the IRGA at a rate of 16.07 L min^{-1} for the bog and 16.73 L min^{-1} for the pasture. Instantaneous measurements of CO_2 and H_2O concentrations were measured inside the sampling cell, along with instantaneous air temperature and air pressure. The enclosed analyzer not only outputs instantaneous gas density for traditional flux calculations (Webb et al., 1980), but also instantaneous mixing ratio of CO_2 and H_2O, which use instantaneous water, temperature and pressure measurements inside the cell to correct for dilution, and thermal and pressure effects, so that post-processing do not need Webb-Pearman-Leuning terms. Two thermocouples were used to measure the instantaneous temperatures of air

just before the air enters the sampling volume and that immediately after the air leaves sampling volume. A differential pressure sensor with a high speed and precision, together with a low speed, high precision absolute pressure sensor was used to measure the instantaneous pressure in the middle of the cell. Data output from the EC systems was recorded at 10 Hz with a data logger (LI-7550, Li-Cor Inc., Nebraska, USA) and stored on a removable USB device. For both sites, a series of environmental variables were recorded by meteorological instruments mounted on the EC system tower. Photosynthetically active photon flux density (PPFD) was measured by quantum sensors facing up and down (LI-190SL-50, Li-Cor Inc., Nebraska, USA) and air temperature (T_a) and relative humidity (RH) were measured with temperature and humidity probes (HMP155, Vaisala, Vantaa, Finland). Four-component net radiometers were used to monitor the incoming and reflected radiation of both short-wave and long-wave (CNR4, Kipp & Zonen, Delf, The Netherlands). A tipping-bucket rain gauge was mounted on the ground to measure total event rainfall (TR-525USW, Texas Electronics, Texas, USA). We measured the soil temperature at the depth of 1 cm, 5 cm, 10cm, 30 cm, 50 cm, and 100 cm (CS616, Campbell Scientific, Utah, USA) and soil moisture at 5 cm, 10 cm, 30 cm and 50 cm (Delta-TML2x, Delta-T Devices, Cambridge, U.K.) at both sites. Water table depth (WT) was monitored by a stainless steel transducer pressure sensor with SDI-12/RS232 connection (CS451, Campbell Scientific, Utah, USA). Rainfall was recorded in 30-min intervals and all the other environmental variables were scanned at 5-s intervals and recorded as half-hourly means by a data logger (CR3000-XT, Campbell Scientific, Utah, USA).

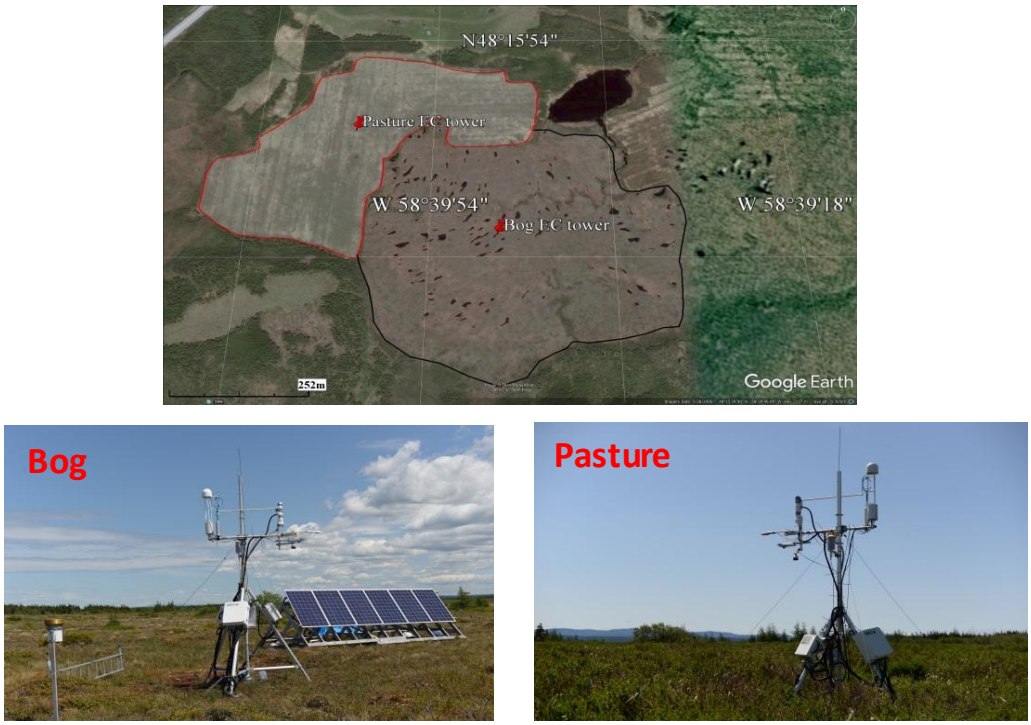


Fig.6.1. The location of flux towers at the bog and abandoned peatland pasture in the Robinsons pasture, western Newfoundland, Canada (bog: 48.260 °N, 58.663 °W; pasture: 48.264 °N, 58.665 °W). The image was from Google Earth with imagery collected on May 28, 2006, with the outline of the site was indicated by the red and black solid line and the red pin represents the location of eddy covariance (EC) tower; one photo of the setup of EC measurement system for the bog and pasture.

6.3.3. EC flux data Processing

EddyPro 5.2.1 software (Li-Cor Inc., Nebraska, USA) was used to process the 10 Hz raw data and output the corrected CO_2, H_2O and CH_4 flux over a 30-min interval. We used the default settings for statistical tests for raw high-frequency data (despiking) (Vickers and Mahrt, 1997), block averaging detrending, correction for frequency response [analytic high-pass filtering correction: (Moncrieff et al., 2004); low-pass filtering correction, selection and configuration:

(Moncrieff et al., 1997)], W-P-L density fluctuations for CH_4 flux (Webb et al., 1980), sonic anemometer tilt correction with double rotation (Wilczak et al., 2001), angle-of-attack correction for wind components (Nakai and Shimoyama, 2012), lag minimization using maximum covariance with default lag of 0, and calculation of friction velocity (u*) using both along and cross wind shear. Footprint lengths were calculated following Kljun *et al.* (2004) and quality flags for the flux calculation were determined following Mauder & Foken (2011). The outputted corrected half-hourly fluxes have diagnostic flags related to the data quality, with the values 0, 1, 2 representing data with good, moderate, and bad quality, respectively. The threshold of u* for CO_2 flux was determined as 0.12 m s^{-1} for the bog and 0.15 m s^{-1} for the abandoned peatland pasture according to the online tool (http://www.bgc-jena.mpg.de/~MDIwork/eddyproc/). We did not find a correlation between CH_4 flux and u*, thus failing to determine a u* threshold. Therefore, we set it as 0.1 m s^{-1} as suggested by a previous study where no u* threshold could be found (Reichstein et al., 2005). The poor quality data with a quality flag of 2, the mean value of received signal strength indicator for LI-7700 less than 20%, u* less than the threshold value, rainfall larger than 0 mm and negative nighttime net ecosystem exchange (NEE) were discarded. Fetches varied in different directions from about 270 to 400 m (0-180: 380 m; 180-360: 300 m) for the bog and between170 and 370 m (0 -45 º: 200 m; 45-77 º: 287 m; 77-115 º: 370 m; 115-160 º: 170 m; 160-250 º: 360 m; 250-360 º: 200 m) for the abandoned peatland pasture. We discarded the flux data with the 70% cumulative footprint larger than these fetches to guarantee the fluxes source are mostly from area within either the bog or pasture boundary. The final flux data were corrected by adding the storage flux value below the height of the EC instruments, which is estimated from temporal changes in gas concentrations measured by either the LI-7200 or LI-7700 and the height integral between the instruments and peatland surface (LI-COR

Biosciences, 2016), under the assumption that flux concentration was invariant with height.

For CO_2 flux, an online tool was used to do the flux gap filling and partitioning (http://www.bgc-jena.mpg.de/~MDIwork/eddyproc/). For CH_4 flux, we employed an artificial neural network (ANN) as proposed by Dengel et al. (2013) to fill the CH_4 flux gaps. We used the neural network Fitting Tool in the mathematical software Matlab to select data, create and train the network, and evaluate its performance using mean square error and regression analysis. The network was trained with a Levenberg-Marquard back-propagation algorithm (trainlm) as used in previous studies (Dengel et al., 2013, Riedmiller, 1994). We chose input variables including air temperature, surface soil temperature, subsurface soil temperature, PPFD, vapor pressure deficit (VPD), u* and WT according to Dengel et al. (2013). However, during some period in wintertime, VPD and u* data were also missing, so we only used the remaining variables mentioned above to fill the data gaps at these times. In order to set a reliable number of neurons in the hidden layer, we applied 1-10 neurons to the standardized approaches (Järvi et al., 2012). The training distribution showed a consistent increase in correlation coefficient with increase in the number of neurons. Therefore, we set the number of neurons in the fitting network's hidden layer as 10. This procedure was replicated for 20 times and the median predictions were used to fill missing half-hour fluxes. Before training, all data were normalized between 0-1 as (Dengel et al., 2013, Moffat et al., 2010) and divided into nighttime and daytime data sets according to a PPFD threshold of 20 $\mu mol\ m^{-2}\ s^{-1}$.

6.3.4. Chamber measurements of N_2O flux

The details of plots set up and chamber flux measurements can be found in our previously published papers (Luan and Wu, 2014, 2015) and we present it briefly here. Three plots,

representing three replicates, were established in both the bog and the abandoned peatland pasture since 2013. For the bog, three subplots in each plot were set up to sample one hummock, hollow and pool. For the abandoned peatland pasture, three subplot were established to sample one drainage ditch and two different plant dominated patches (lower herbaceous and graminoid dominated and shrubs, e.g. sweet gale, Labrador tea dominated). We inserted PVC (Polyvinyl chloride) collars with an inner diameter of 26 cm into the peat to a depth of 10 cm permanently in each subplot in May 2013 two weeks before the start of our measurements. For each pool, four ABS (Acrylonitrile butadiene styrene) pipes were inserted into the pool bottom to hold the chamber on the water to make the floating chamber measurement possible. The air sample was taken from the opaque PVC chamber with an inner diameter of 26 cm placed on the collar with a water seal. Four air samples were taken into 30 ml plastic syringes equipped with a three-way stopcock through the rubber septum over the 30 minutes' incubation period at an interval of 10 minutes. The gas samples were analyzed using a Bruker gas chromatograph equipped with an electron capture detector (Nykanen *et al.*, 1995), where two concentrations of standard gases were employed to calibrate our air samples to calculate the concentration of N_2O. The gas flux rate was calculated from the linear change in concentration as: $F = (dC/dt) \times V/A$ (where V is the volume of the chamber, A is the chamber cover area, and dC/dt is the rate of concentration change). Only data with the regression r^2 greater than 80% were used for calculating the N_2O flux. The sampling was conducted during the growing season from 2013 to 2016 at an interval of two-four weeks. We also measured the soil temperature at 5 cm and 20 cm and water table depth and collected soil water samples at 10 cm and 40 cm below surface to analyze the dissolved organic carbon (DOC), total organic carbon (TOC) and total nitrogen (TN) concentrations when taking the air samples. Cumulative growing season fluxes of N_2O was obtained through linear

interpolation of biweekly/monthly static chamber measurements (Luan *et al.*, 2015, Teh *et al.*, 2011). Spatial weightings were applied to each of the three communities based on their respective areal fractions as follows, weighted mean = $\Sigma f_i X_i / \Sigma f_i$, where X_i is the flux for the given land forms, and f_i is the percentage of coverage corresponding to the given land forms (Luan et al., 2015, Teh et al., 2011). Hummocks, hollows and pools occupy approximately 45%, 45% and 10% of the bog, respectively; grass-dominated communities, shrub-dominated communities and ditches account for nearly 45%, 45% and 10% of the pasture, respectively.

6.3.5. Net global warming potential and sustained-flux global warming/cooling potentials (SGWP, SGCP) calculations

For decades, ecosystem scientists have used GWP to compare the radiative forcing of various greenhouse gases to determine if the ecosystems have a net warming or cooling effect on climate. The use of GWP requires the implicit assumption that greenhouse gas emissions occur as a single pulse, but this assumption is rarely justified in ecosystem studies. Two alternative metrics of the SGWP (for gas emissions) and SGCP (for gas uptake) have been recently presented to describe the radiative forcing of greenhouse gases that persist over time (Neubauer and Megonigal, 2015). In this study, we used both GWP and the new metrics (SGWP/SGCP) to describe the warming/cooling functions of the greenhouse gases. The reason for the use of GWP in this study is to better compare the climate warming/cooling potential with other peatlands that reported GWP values. On a 100-year time horizon, the GWP, SGWP and SGCP of CH_4 flux are 25, 45 and 203 times that of CO_2; the GWP, SGWP and SGCP of N_2O flux are 298, 270 and 349 times that of CO_2 (IPCC, 2014; Neubauer and Megonigal, 2015). The net flux data of CH_4 and N_2O were converted to equivalent CO_2 units in order to compare the total GWP and SGWP of all

three greenhouse gases. Therefore, GWP and SGWP/SWCP flux refers to the net flux of CO_2, CH_4 and N_2O expressed in equivalent CO_2 units (IPCC, 2014; Neubauer and Megonigal, 2015).

6.4. Results

6.4.1. Environmental variables

For both the bog and the abandoned peatland pasture, air temperature was higher than normal in July 2014 and September 2015 and colder than normal conditions in March, April, June, July and November 2015 and April 2016 (Table 6.1). The rainfall was lower than normal in September 2015, but higher than normal rainfall occurred in three consecutive winter months from November 2014 to January 2015 (Table 6.1). In all other months temperature and precipitation were within ± one standard deviation of the long-term normal. Environmental variables including air and soil temperature, PPFD and rainfall were very similar in the bog and the abandoned peatland pasture (Fig.6.2, Table 6.2). However, the water table averaging -42 cm in 2014 and ~-30 cm in 2015 at the abandoned peatland pasture were lower than that of -21 cm in 2014 and -23 cm in 2015 at the bog, respectively (Fig.6.2d, Table 6.2). Soil water content was between 0.61 and 0.67 m^3 m^{-3} at the abandoned peatland pasture throughout the whole study period, which was lower than that of 0.76 -0.88 m^3 m^{-3} at the bog (Fig.6.2: c1-c3).

Table 6.1. Comparison of monthly average temperature and cumulative monthly rainfall measured at Robinson Pasture during measurement periods from April, 2014 to May, 2016 with the long-term (1981-2010 average ± SD) measurements from the nearby, climate station in Stephenville, Newfoundland and Labrador. Please note that bog and pasture had similar rainfall.

Month	Rainfall (mm)				Air temperature (℃)						
	Bog & Pasture				Bog	Pasture	Bog	Pasture	Bog	Pasture	
	2014	2015	2016	1981-2010	2014		2015		2016		1981-2010
Jan		54	14	29 ±24			-6.7	-6.8	-5.2	-5.3	-6 ±1.6
Feb		20	45	27 ±30			-9.2	-9.2	-3.9	-3.9	-6.7 ±2.9
Mar		12	30	37 ±29			-6.9	-6.9	-5	-5	-3.5 ±2.5
Apr		41	88	62 ±42			-0.8	-0.8	0.6	0.6	2.6 ±1.8
May	129	118	106	94 ±44	6.5	6.5	7	7.1		7.4	7.6 ±1.4
Jun	65	64		104 ±45	12.3	12.2	10.1	10.2			12.1 ±1.3
Jul	97	119		118 ±45	19.3	19	14.2	14.1			16.4 ±1.1
Aug	105	125		130 ±65	16.7	16.5	18.1	17.9			16.7 ±0.9
Sep	83	55		128 ±48	12.4	12.2	13.8	13.7			12.8 ±1.1
Oct	85	101		124 ±45	8.6	8.5	6.5	6.4			7.4 ±1.3
Nov	133	82		94 ±31	1.6	1.5	1.2	1.2			2.7 ±1.3
Dec	105	54		49 ±42	-1.4	-1.5	-2.1	-2.1			-2.4 ±1.7
Overall		845		995 ±133			3.7	3.7			5.0 ±1

Table 6.2. Comparison of average daily air temperature, soil temperature at depth of 1cm and 50 cm, photosynthetic photon flux density (PPFD), cumulative rainfall, and water table for different periods in the two study years at both bog and pasture

Period	Study year	Air temperature (℃)		Soil temperature (℃)				PPFD		Water table		Rainfall
				1 cm		50 cm		(mol m^{-2} d^{-1})		(cm)		(mm)
		Bog	Pasture	Bog	Pasture	Bog	Pasture	Bog	Pasture	Bog	Pasture	Bog/ Pasture
Growing season	2014-15	13.1	12.8	13.7	13.1	10.4	10.5	29.7	28.7	-21	-42	513
	2015-16	11.5	11.4	12.3	11.8	10	9.9	25.1	25.4	-23	-29	603
Non-growing season	2014-15	-5.6	-5.8	-0.4	0.3	2.1	2.4	19.7	18.2	-17	-17	126
	2015-16	-3.4	-3.4	-0.1	0	1.9	2.3	15.5	15.1	-22	-11	174
Overall	2014-15	4.4	4.3	6.9	6.9	6.7	6.7	22.7	22.7	-18	-28	936
	2015-16	4.7	4.6	6.4	6.3	6.3	6.5	20	20	-22	-21	890

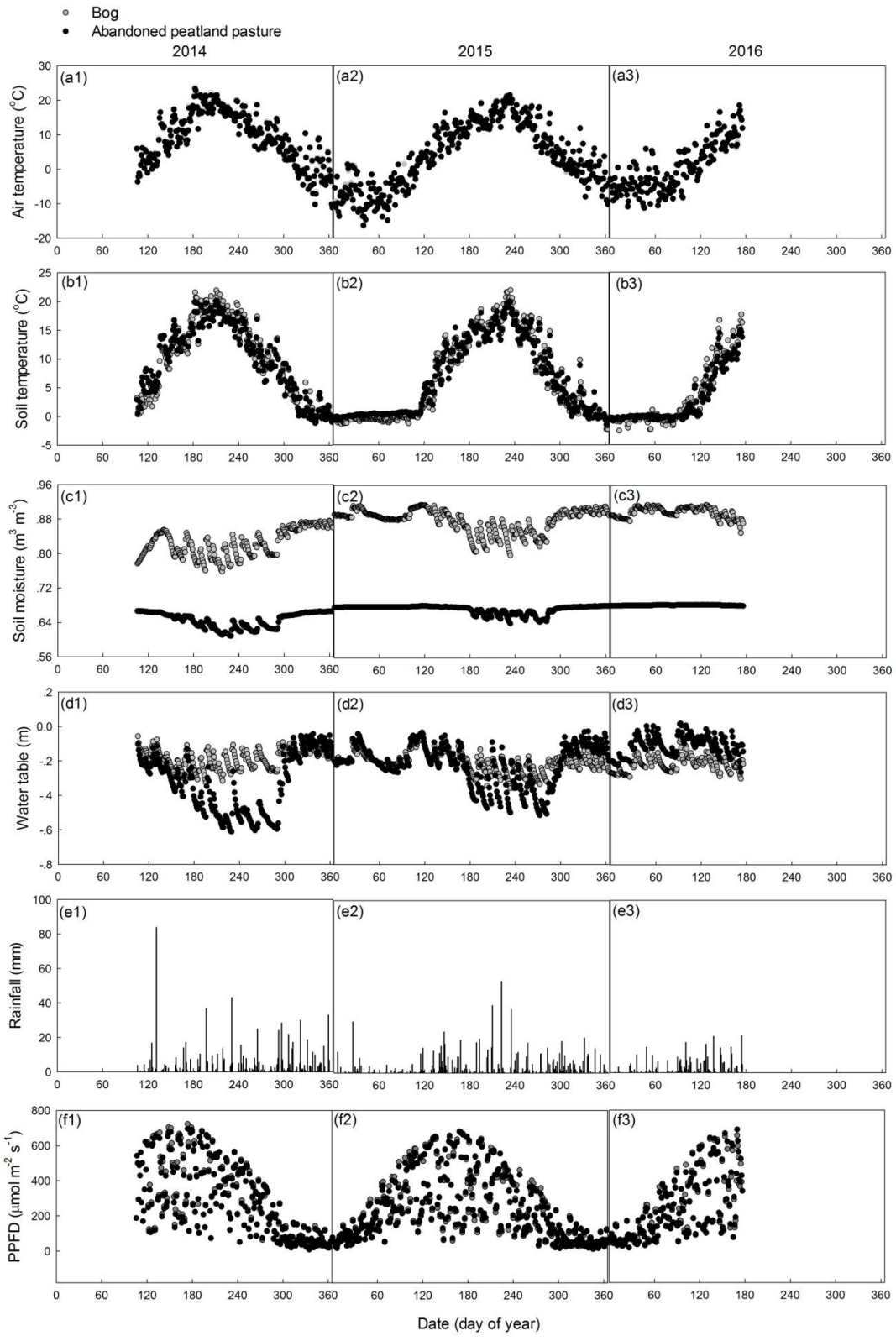

Bog
Abandoned peatland pasture
2014
2015
2016
(a1)
(a2)
(a3)
Air temperature (°C)
(b1)
(b2)
(b3)
Soil temperature (°C)
(c1)
(c2)
(c3)
Soil moisture (m³ m⁻³)
(d1)
(d2)
(d3)
Water table (m)
(e1)
(e2)
(e3)
Rainfall (mm)
(f1)
(f2)
(f3)
PPFD (μmol m⁻² s⁻¹)
Date (day of year)

Fig.6.2. The daily average air temperature (a1-a3), soil temperature at 1 cm (b1-b3), volumetric soil water content at depth of 30 cm (c1-c3), water table (d1-d3), cumulative rainfall (e1-e3) and photosynthetic photon flux density (PPFD) (f1-f3) during the measurement periods for both the bog and the abandoned peatland pasture.

6.4.2. Comparison of CO_2 fluxes and annual budgets between the two sites

The general pattern of NEE followed a similar seasonal pattern at both the bog and the abandoned peatland pasture, with most net CO_2 uptake occurring from late May to September (Fig.6.3a). For the bog, the highest daily average net CO_2 uptake rate of -2.61 μmol m^{-2} s^{-1} occurred in the early August of 2014 and -1.84 μmol m^{-2} s^{-1} in the middle July of 2015 (Fig.6.3a). In contrast maximum NEE uptake at the abandoned peatland pasture was significantly larger (in an absolute sense) at -5.88 μmol m^{-2} s^{-1} in 2014 and -6.38 μmol m^{-2} s^{-1} in 2015, which occurred in early July of both years (Fig.6.3a). Daily net CO_2 emissions for both sites were found during the growing season on days with low PPFD but high air and soil temperature. ER of both sites showed similar seasonal patterns, values increased from the early growing season and peaked in the end of July in 2014 and in the end of August in 2015, being consistent with the maximum soil temperature at 1 cm (Fig.6.3b). Maximum ER was 8.64 μmol m^{-2} s^{-1} in 2014 and 8.43 μmol m^{-2} s^{-1} in 2015 for the abandoned peatland pasture, two to three times higher than for the bog (3.40 μmol m^{-2} s^{-1} in 2014 and 3.33 μmol m^{-2} s^{-1} in 2015). GPP increased from near zero in the early growing season and reached up to 13.44 μmol m^{-2} s^{-1} for both years at the abandoned peatland pasture and 4.92 μmol m^{-2} s^{-1} in 2014 and 4.52 μmol m^{-2} s^{-1} in 2015 at the bog, occurring in the early-middle July for the pasture, but in the early August in 2014 and middle July in 2015 for the bog (Fig.6.3c). The maximum GPP of the pasture was about three times that of the bog.

The bog accumulated 109 g CO_2-C m^{-2} yr^{-1} in 2014-15 during the growing season in 2014-15, 83% of which was offset by the non-growing season emissions, making it a small CO_2 sink of 18 g CO_2-C m^{-2} yr^{-1} in 2014-15 (Table 6.3). For the pasture, although the growing season CO_2 uptake of 213 g CO_2-C m^{-2} yr^{-1} in 2014-15 was higher than that of 182 g CO_2-C m^{-2} yr^{-1} in 2015-16, the annual CO_2 uptake of -85 g CO_2-C m^{-2} yr^{-1} in 2014-15 was similar to that of -82 g CO_2-C m^{-2} yr^{-1} in 2015-16, mainly due to its higher low non-growing season CO_2 emission (Table 6.3).

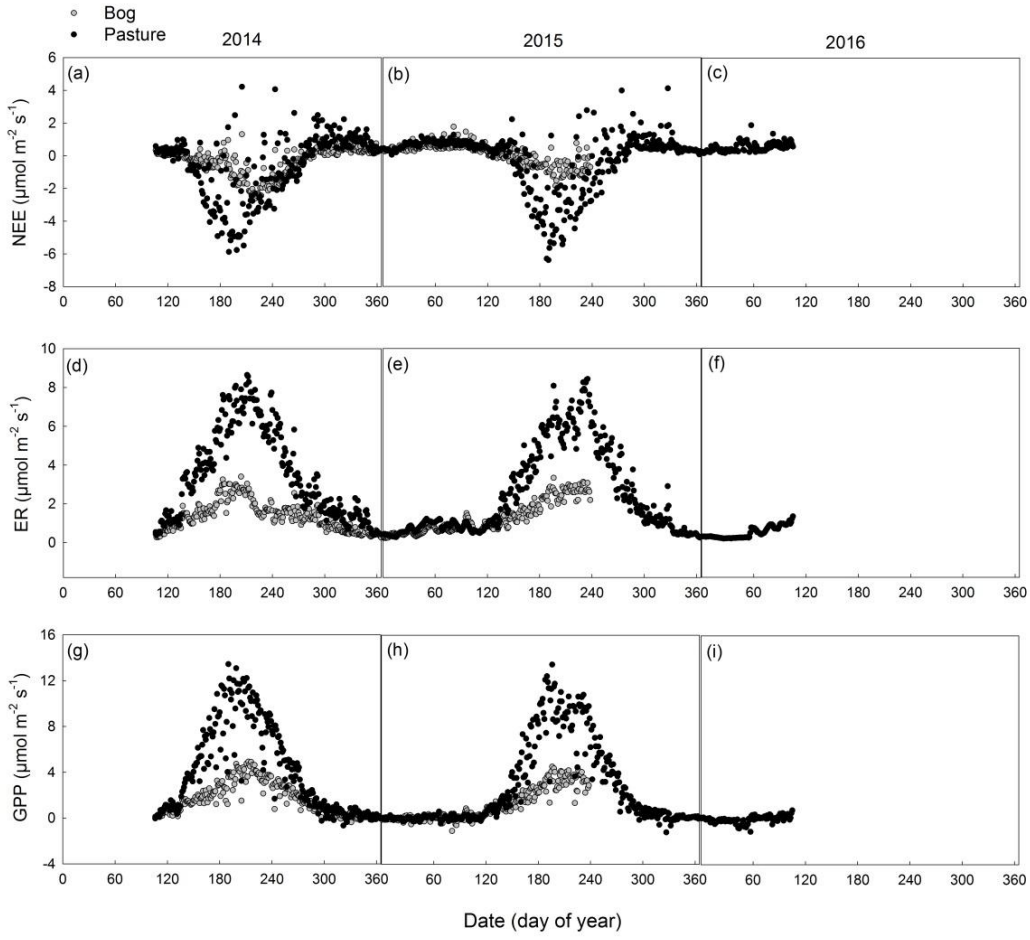


Fig.6.3. The daily variation in net ecosystem exchange (NEE) (a1-a3), ecosystem respiration (ER) (b1-b3) and gross primary production (GPP) (c1-c3) during the period between April 2014 and August 2015 at the bog and between April 2014 to April 2016 at the abandoned peatland pasture.

Table 6.3. The total accumulated fluxes of CO_2 , CH_4 and N_2O, carbon balance, the total global warming potential (GWP) and sustained-flux global warming potential (SGWP) for the growing season, non-growing period and the annual budget in two years from April 2014 to April 2016 for both bog and pasture. Note that CO_2 flux values between September 2015 and April 2016 were derived based on the relationship between the monthly average CO_2 flux and air temperature, and N_2O flux was not available in the non-growing season at both sites.

Site	Period	CO_2 flux (g CO_2-C m^{-2})		CH_4 flux (g CH_4-C m^{-2})		N_2Oflux (g N_2O-N m^{-2})		Carbon balance (g C m^{-2} yr^{-1})		GWP (g CO_2-eq m^{-2})		SGWP (g CO_2-eq m^{-2})	
		2014-15	2015-16	2014-15	2015-16	2014-15	2015-16	2014-15	2015-16	2014-15	2015-16	2014-15	2015-16
Bog	Growing season	-99 ±57	-66 ±38	1.88 ±0.66	2.04 ±0.56	-0.050 ±0.036	-0.066 ±0.193	-97 ±57	-64 ±38	-91 ±61	6 ±70	-49 ±29	-15 ±175
	Non-growing season	80 ±93	67 ±77	1.70 ±0.20	1.03 ±0.37	n/a	n/a	82 ±93	68 ±77	123 ±93	93 ±78	137 ±104	101 ±77
	Annual	-18 ±14	1 ±1	3.58 ±0.69	3.07 ±0.67	n/a	n/a	-14 ±11	4 ±1	32 ±23	99 ±62	88 ±63	86 ±61
Pasture	Growing season	-215 ±94	-182 ±70	0.13 ±0.19	0.20 ±0.24	-0.004 ±0.045	-0.080 ±0.199	-215 ±94	-182 ±70	-213 ±95	-201 ±92	-211 ±94	-197 ±92
	Non-growing season	130 ±117	108 ±122	0.14 ±0.12	-0.11 ±0.14	n/a	n/a	130 ±117	108 ±122	134 ±117	105 ±122	135 ±117	91 ±105
	Annual	-85 ±40	-82 ±37	0.27 ±0.23	0.10 ±0.29	n/a	n/a	-85 ±40	-82 ±37	-78 ±40	-96 ±38	-76 ±39	-106 ±50

6.4.3. Temporal variability in CH_4 and N_2O fluxes and budgets

For the bog strong seasonal variability in the daily average CH_4 fluxes was observed in both 2014 and 2015 (Fig.6.4). The CH_4 emission rates ranged from near zero in early May to a peak of above 20 nmol m^{-2} s^{-1} occurring in the middle-late August in 2014 and in the early-middle September in 2015 and then decreased to near zero by the end of the growing season in both years. During the non-growing season periods, CH_4 fluxes were quite noisy, ranging from approximately -10 to 20 nmol m^{-2} s^{-1}. For the pasture, the seasonal pattern of CH_4 fluxes was not strong in either study year, though some pronounced periods of emissions occurred in 2014/2015 winter and late growing season in 2015, but uptake occurred in all season in both years (Fig.6.4). The CH_4 fluxes were small varying around the zero line, with the daily average CH_4 flux ranging from -4.1 to 9.9 nmol m^{-2} s^{-1} over the first study year and from -7.1 to 12.1 nmol m^{-2} s^{-1} over the second study year (Fig.6.4). The accumulated annual CH_4 flux of ~2.7 g CH_4-C m^{-2} yr^{-1} in 2014-15 and 2.3 g CH_4-C m^{-2} yr^{-1} in 2015-16, for the bog was higher than that of 0.27 ± 0.23 g CH_4-C m^{-2} yr^{-1} and 0.10 ± 0.29 g CH_4-C m^{-2} yr^{-1} for the abandoned peatland pasture, respectively (Table 6.3).

N_2O flux showed little temporal variability for each landform of both the bog and the pasture (Fig.6.5). The growing season N_2O flux was not significantly different from zero for most landforms of both sites in the four study years, ranging from -0.70 ± 1.19 mg m^{-2} d^{-1} to 0.21 ± 0.24 mg m^{-2} d^{-1} (Fig.6.6, Table 6.4), with the accumulated growing season N_2O flux of -0.458 ± 0.51 g N_2O -N m^{-2} in 2014-15 and -0.066 ± 0.193 g N_2O m^{-2} in 2015-16 at the bog and -0.004 ± 0.045 g N_2O m^{-2} in 2014-15 and -0.080 ± 0.199 g N_2O m^{-2} in 2015-16 at the abandoned peatland pasture (Table 6.3).

6.4.4. Annual C and greenhouse gases budget

Both the bog and the abandoned peatland pasture acted as C sink when considering the flux of CO_2 and CH_4 flux, with a net C accumulation of ~ -14 ± 11 g C m^{-2} yr^{-1} in 2014-15 for the bog and ~ -85 ± 40 g C m^{-2} yr^{-1} in 2014-15 and -82 ± 37 g C m^{-2} yr^{-1} in 2015-16 for the abandoned peatland pasture (Table 6.3). The abandoned peatland pasture functioned as "climate cooling" with GWP of -78 ± 40 g CO_2-C equivalents m^{-2} yr^{-1} in 2014-15 and -96 ± 38 g CO_2-C equivalents m^{-2} yr^{-1} in 2015-16 and SGWP of -76 ± 38 g CO_2-C equivalents m^{-2} yr^{-1} in 2014-15 and -106 ± 50 g CO_2-C equivalents m^{-2} yr^{-1} in 2015-16 (Table 6.3). However, the bog acted as climate warming, with GWP of 32 ± 23 g CO_2-C equivalents m^{-2} yr^{-1} in 2014-15 and 99 ± 62 g CO_2-C equivalents m^{-2} yr^{-1} in 2015-16 and SGWP of 88 ± 63 g CO_2-C equivalents m^{-2} yr^{-1} in 2014-15 and 86 ± 61 g CO_2-C equivalents m^{-2} yr^{-1} in 2015-16 (Table 6.3). It suggested the global "climate cooling" potential of the abandoned peatland pasture was more than 150 g CO_2-C equivalents m^{-2} yr^{-1} that of the bog.

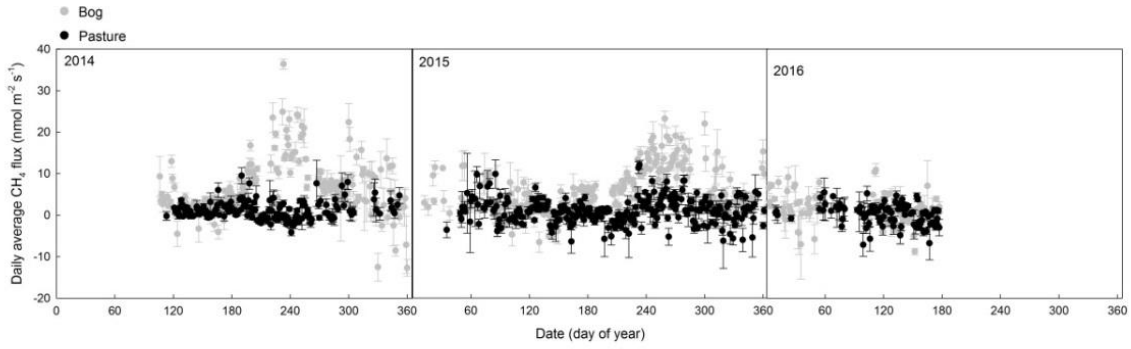


Fig.6.4. Comparison of the daily variation in CH_4 flux between the bog and the abandoned peatland pasture from April 2014 to April 2016. Points are daily average flux and error bar indicates standard error of the average.

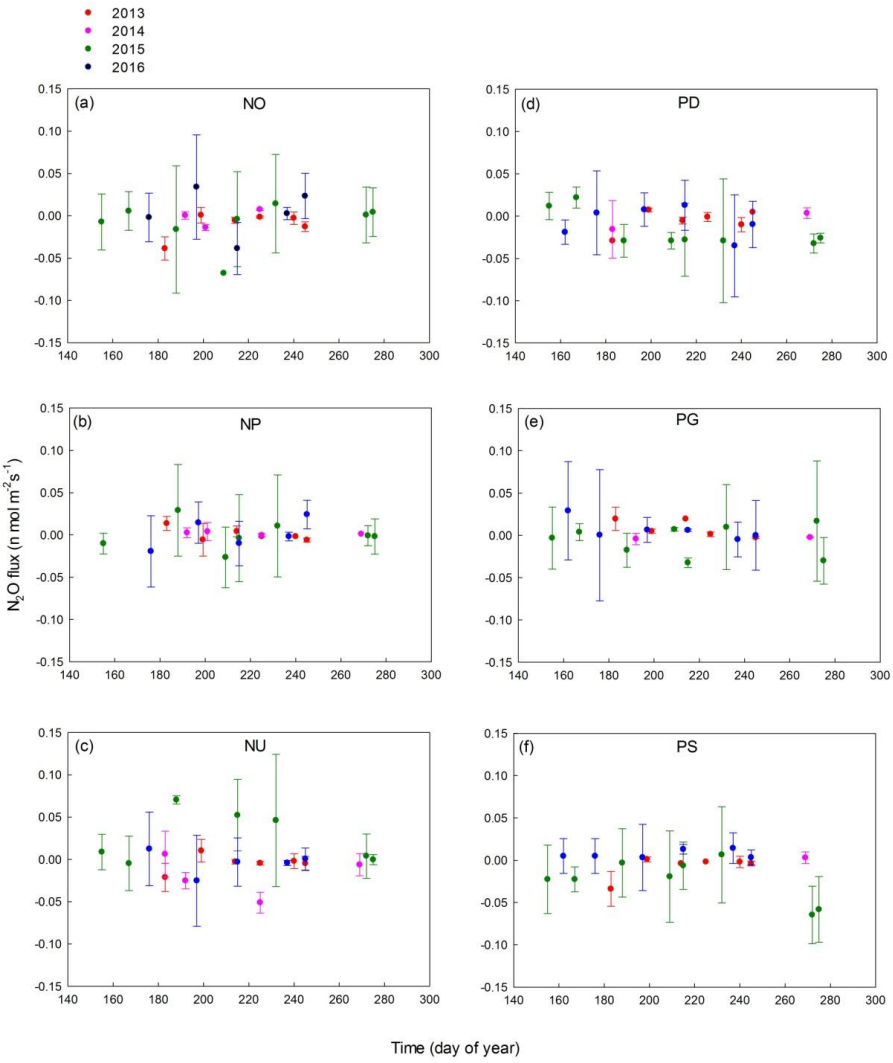


Fig. 6.5. The temporal variation in N_2O flux for different landforms of both the bog and pasture. NO, NU and NP indicate the hollow, hummock and pool in the bog, respectively, and PG, PS and PD represent grass-dominated patch, shrub-dominated patch and ditch in the pasture, respectively.

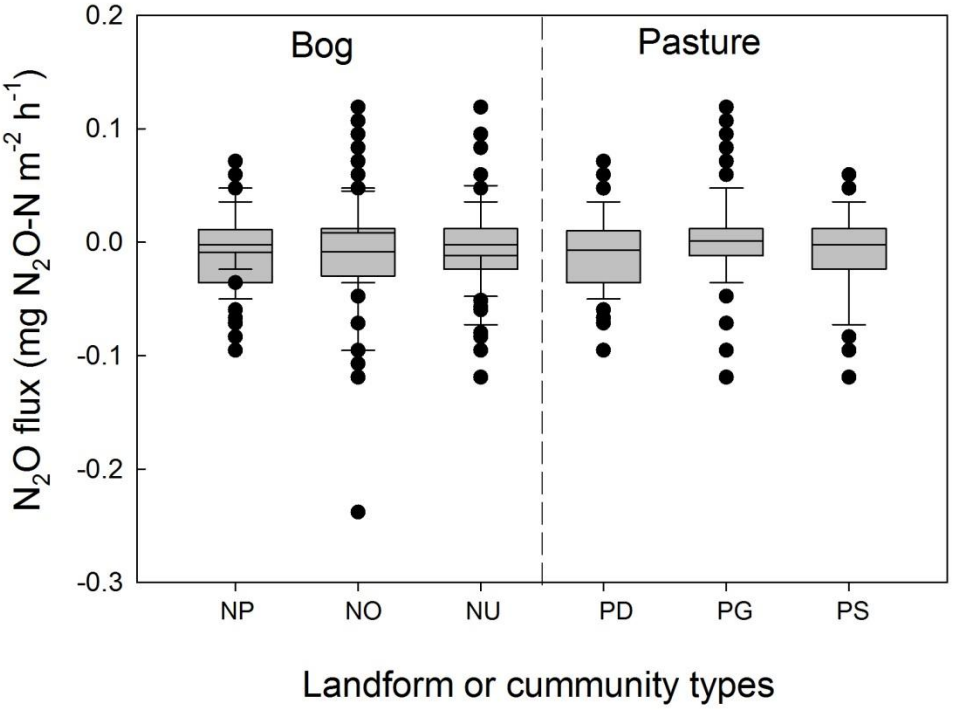


Fig.6.6. Comparison of the mean growing season N_2O flux in different landforms of the bog and abandoned peatland pasture. The middle black solid line indicates the median value.

Table 6.3. The total accumulated fluxes of CO_2 , CH_4 and N_2O, carbon balance, the total global warming potential (GWP) and sustained-flux global warming potential (SGWP) for the growing season, non-growing period and the annual budget in two years from April 2014 to April 2016 for both bog and pasture. Note that CO_2 flux values between September 2015 and April 2016 were derived based on the relationship between the monthly average CO_2 flux and air temperature, and N_2O flux was not available in the non-growing season at both sites.

Site	Period	CO_2 flux (g CO_2-C m^{-2})		CH_4 flux (g CH_4-C m^{-2})		N_2O flux (g N_2O-N m^{-2})		Carbon balance (g C m^{-2} yr^{-1})		GWP (g CO_2-eq m^{-2})		SGWP (g CO_2-eq m^{-2})	
		2014-15	2015-16	2014-15	2015-16	2014-15	2015-16	2014-15	2015-16	2014-15	2015-16	2014-15	2015-16
Bog	Growing season	-99 ±57	-66 ±38	1.88 ± 0.66	2.04 ±0.56	-0.050 ± 0.036	-0.066 ± 0.193	-97 ±57	-64 ±38	-91 ±61	6 ±70	-49 ±29	-15 ±175
	Non-growing season	80 ±93	67 ±77	1.70 ± 0.20	1.03 ±0.37	n/a	n/a	82 ±93	68 ±77	123 ±93	93 ±78	137 ±104	101 ±77
	Annual	-18 ±14	1 ±1	3.58 ± 0.69	3.07 ±0.67	n/a	n/a	-14 ±11	4 ±1	32 ±23	99 ±62	88 ±63	86 ±61
Pasture	Growing season	-215 ±94	-182 ±70	0.13 ± 0.19	0.20 ±0.24	-0.004 ± 0.045	-0.080 ± 0.199	-215 ±94	-182 ±70	-213 ±95	-201 ±92	-211 ±94	-197 ±92
	Non-growing season	130 ±117	108 ±122	0.14 ± 0.12	-0.11 ± 0.14	n/a	n/a	130 ±117	108 ±122	134 ±117	105 ±122	135 ±117	91 ±105
	Annual	-85 ±40	-82 ±37	0.27 ± 0.23	0.10 ±0.29	n/a	n/a	-85 ±40	-82 ±37	-78 ±40	-96 ±38	-76 ±39	-106 ±50

Table 6.4. The N_2O flux (average ± standard error) of different landforms in both bog and pasture during the growing season of 2013-16.

Site	Landform	N_2O flux (mg N_2O-N m^{-2} d^{-1})			
		2013	2014	2015	2016
Bog	Hollow	-0.30 ± 0.23	-0.06 ± 0.10	0.11 ± 1.60	0.11 ± 0.60
	Pool	0.12 ± 0.37	0.05 ± 0.18	-0.24 ± 1.03	-0.07 ± 0.84
	Hummock	-0.13 ± 0.30	-0.52 ± 0.48	0.77 ± 0.55	-0.09 ± 0.80
Pasture	Ditch	-0.16 ± 0.16	-0.18 ± 0.92	-0.51 ± 0.95	-0.19 ± 1.10
	Grass-dominated patch	-0.24 ± 0.54	-0.10 ± 0.14	-0.17 ± 1.05	0.18 ± 1.30
	Shrub-dominated patch	-0.16 ± 0.33	0.09 ± 0.20	-0.70 ± 1.19	0.21 ± 0.64

6.5. Discussion

6.5.1. Comparison of CO_2 fluxes between the bog and pasture

The agriculturally drained peatlands have been mostly reported to be large CO_2 sources due to high rates of peat oxidation under low water table conditions (Knox et al., 2015; Nieveen et al., 2005), motivating great efforts to mitigate their CO_2 emissions. However, our results indicated that this abandoned peatland pasture was a stronger CO_2 sink than that for the undisturbed bog, which represents the natural condition from which the pasture was originally developed. Annual NEE for the abandoned peatland pasture was about five times that of the bog. Although both component fluxes (GPP and ER) were larger at the abandoned peatland pasture, its higher net CO_2 uptake capacity was due to a larger difference in GPP, which reflects its higher productivity, with its maximum AGB ~3 times that of the adjacent bog. Similarly, a few previous studies suggested that limited emissions or even uptakes of CO_2 occurred in agriculturally managed peatlands under low intensity of land use (Hendriks et al., 2007; Wilson et al., 2016), and removing grazing on the drained peatlands can return their C sequestration function (Wilson et al., 2016). Traditionally, rewetting and restoring the agriculturally managed peatlands has been

found to be a promising way to reverse their CO_2 losses since it can effectively stop peat oxidation (Günther et al., 2015; Karki et al., 2016; Knox et al., 2015; Soini et al., 2010; Strack et al., 2014). Here, we find that the CO_2 sequestration capacity of the abandoned peatland pasture was comparable to those restored peatlands, implying that the abandonment for degraded managed peatlands to let them develop naturally can be as efficient as other widely used restoration approaches to enhance their CO_2 sequestration capacity.

6.5.2. Comparison of CH_4 fluxes between the bog and pasture

The CH_4 emissions rates of both sites were within the previously reported flux ranges for pristine (Long et al., 2010; Roulet et al., 2007; Salm et al., 2012; Shannon and White, 1994) and drained peatlands (Kasimir Klemedtsson et al., 2009; Kroon et al., 2010; Maljanen et al., 2010), respectively. The annual CH_4 emission rate was ~0.1-0.3 g CH_4 m^{-2} yr^{-1} in the abandoned peatland pasture, lower than that of 3.1-3.6 g CH_4 m^{-2} yr^{-1} at the bog, consistent with previous findings that the CH_4 emissions can be significantly reduced in drained peatlands (Maljanen et al., 2010). However, the restored peatlands have shown high CH_4 emission rates, ranging from ~7 to 63 g CH_4 m^{-2} yr^{-1}, due to high WT after rewetting (Günther et al., 2015, Karki et al., 2016; Waddington and Day, 2007; Wilson et al., 2016). This implies that one advantage of the abandoned peatland pasture over the rewetted and restored peatlands lies in its much lower CH_4 emissions. Numerous studies have shown a strong relationship between NEE or plant productivity and CH_4 flux, as vegetation is the primary source of C substrate for methanogenic metabolism (Hatala et al., 2012; Whalen, 2005; Whiting and Chanton, 1993). Indeed, we found the daily average CH_4 flux increased with enhanced daily net CO_2 uptake rate when daily average NEE was negative for the bog (unpublished results). However, the cumulative CH_4 flux during the growing season was lower in 2014 than in 2015, though the CO_2 uptake rate was

higher in 2014 than that in 2015 for the abandoned peatland pasture. The higher overall growing season CH_4 flux in 2015 was due to higher WT, suggesting that WT was the main control over the variation in growing season cumulative CH_4 flux in the abandoned peatland pasture.

6.5.3. Comparison of N_2O fluxes between the bog and pasture

We did not find strong temporal and spatial variation in N_2O flux in different landforms of both the bog and the abandoned peatland pasture since the mean emission rates were mostly not significantly different from zero, suggesting that N_2O flux can be ignored when estimating the annual GHGs balance in both sites. Typically, N_2O flux should be low in pristine peatlands with low nutrient condition (Martikainen et al., 1993; Nykanen et al., 1995; Regina et al., 1996) and most previous studies have not considered N_2O flux when studying the role of natural peatlands in climate change. Agricultural management should increase N_2O emission of peatlands because manure application and drier conditions can promote N_2O production (Haddaway et al., 2014). Large N_2O emissions were observed in agriculturally managed peatlands elsewhere with the emission ranging from 0.2-1.6 g N m^{-2} yr^{-1} (Maljanen et al., 2010; Nykanen et al., 1995; Salm et al., 2009; Teh et al., 2011). However, the N_2O flux from this abandoned peatland pasture was insignificant, similar to an abandoned peatland meadow (Hendriks et al., 2007) and a restored managed peatland (Mander, et al., 2012), a rewetted temperate fen (Günther et al., 2015) and a disturbed bog at different stages of recovery in British Columbia (Christen et al., 2016). Although the TN concentration in the pasture (1.24 - 2.26 mg L^{-1}) was higher than that in the bog (0.74 - 0.93 mg L^{-1}) (Table 6.5), we did not find significant difference in the N_2O flux between the sites, probably due to the simulation of plant growth under high N condition. The enhancement of plant growth can compete for NO_3^- with denitrifying microbes and moderate the N_2O emissions (Silvan et al., 2005). In addition, the insignificant N_2O flux in the pasture may

partly because the increase magnitude of N availability was too small to promote N_2O emission. The TN concentration in our pasture (1.24-2.26 mg L^{-1}) is low when compared to that in other managed peatlands (above 4 mg L^{-1}) (Harpenslager, et al., 2015; Tiemeyer, et al., 2007), probably due to its long-term abandonment of more than 25 years. N_2O emission was found to be promoted only under condition of very high N availability, with the simulation of N_2O emissions occurring only when N application were very high (480 kg N ha^{-1}), but not found under a moderate N supply (Augustin et al. 1998). Similarly, drainage has been suggested to increase N_2O flux from nutrient-rich sites, but has little effect at nutrient-poor site (Martikainen et al., 1993).

6.5.4. Comparison of C and GHGs balance between the bog and pasture

The C balance of the bog was primarily determined by the magnitude and direction of CO_2 flux, but the GHG balance was mainly determined by the CH_4 flux. The bog was a small C sink of ~ -14 g C m^{-2} yr^{-1} in 2014-15 and acted as global warming with GWP of 32 g CO_2-C equivalents m^{-2} yr^{-1} and SGWP of 88 g CO_2-C equivalents m^{-2} yr^{-1} in 2014-15, both of which were within the previous reported range of values (Blais et al., 2005; Levy and Gray, 2015; Nilsson et al., 2008; Nykanen et al., 1995; Pullens et al., 2016; Roulet et al., 2007). For the abandoned peatland pasture, CO_2 flux was found to be the main component of both C and GHG balances, with its GWPs/SGWPs balance of -78/-76 g CO_2-C equivalents m^{-2} yr^{-1} in 2014-15 and -96/-106 g CO_2-C equivalents m^{-2} yr^{-1} in 2015-16. This net GHGs sink of this abandoned peatland pasture was larger than that at an abandoned peatland meadow in the Netherlands where the GWPs balance was -23 g CO_2-C equivalents m^{-2} yr^{-1} (Hendriks et al., 2007). The difference mainly originated from its higher CH_4 emission due to rewetting in the latter site (Schrier-Uijl et al., 2014). For other drained and agriculturally managed peatlands, they have been found to be large GHG

sources, with the GWP ranging from 100 to 3140 g CO_2-equivalents m^{-2} yr^{-1}, mainly due to large CO_2 and N_2O emissions (Frolking and Roulet, 2007; Hendriks et al., 2007; Maljanen et al., 2010; Strack and Waddington, 2007; Teh et al., 2011).

6.5.5. Comparison of C and GHGs budgets between the pasture and restored peatlands elsewhere

In the context of global warming, restoration of these degraded peatlands to mitigate their climate warming potential has attracted much research interest (Couwenberg, 2011; Hendriks et al., 2007). Paludiculture, with cultivation with high productive perennial bioenergy crop (reed canary grass: RCG, *Phalaris arundinaceae*, L.) on drained or rewetted peatlands, is one acknowledged possible land use option on rewetted organic soil, since it avoids the problem of GHG emissions from drained peatlands without scarifying the biomass production of crops (Don et al., 2012; Schröder et al., 2015). Previous studies suggested that the GHGs emissions of drained peatlands after rewetting and crop (RCG) cultivation ranged from 30 to 810 g CO_2-equivalents m^{-2} yr^{-1} (Järveoja et al., 2015; Kandel et al., 2016; Karki et al., 2014, 2015, 2016), comparable to the values of pristine peatlands but lower than that of 2920 ± 1740 g CO_2-equivalents m^{-2} yr^{-1} for drained peatlands (Tiemeyer et al., 2016). For our pasture RGC is one of the dominant species, making it comparable to those restored peatlands under paludiculture. We found that the abandoned peatland pasture was an even stronger GHGs sink (GWP: -78~-96 g CO_2-equivalents m^{-2} yr^{-1}) than most restored peatlands which are GHGs sources (GWP: 30-810 g CO_2-equivalents m^{-2} yr^{-1}), primarily due to either less CH_4 or N_2O emissions in our pasture than those restored counterparts. This implies that abandonment on agriculturally managed peatlands can improve the C sequestration capacity and climate cooling function above the level of its natural state.

Chapter 7-Summary and Conclusion

One objective in this thesis was to compare the temporal variations in CO_2 and CH_4 fluxes and the controls based on continuous EC measurements. In chapter 3 (paper 1) and chapter 4 (paper 2), the temporal patterns of CH_4 flux were found to be different between the bog and abandoned peatland pasture. Bog showed a distinct diel pattern of CH_4 flux during the growing season, with higher CH_4 emissions in the daytime but lower emissions in the nighttime. However, no obvious diel pattern was observed at the abandoned peatland pasture. The diel pattern of CH_4 was found to be positively related to PPFD and this study speculated the underlying mechanisms were related to plant-mediated convective flow or diffusion at the bog. However, we still lack clear understanding about the real underlying mechanisms since many processes operate at the same time in the field.

The two sites also showed very different seasonal patterns of CH_4 flux and so did the controls. At the bog, strong seasonal variability in the daily average CH_4 fluxes was observed in both 2014 and 2015, ranging from near zero before May to a peak of above 20 nmol m^{-2} s^{-1} in the middle-late August in 2014 and in the early-middle September in 2015. At the abandoned peatland pasture, the seasonal variation in daily CH_4 flux was not strong in the two study years, however a few periods of pronounced emissions occurred in wintertime (2014-15) and the late growing season (2015-16). T_{50} was the dominant control on the seasonal variation in the daily average CH_4 flux during the growing season and u* was the main one in the non-growing season in both years at the bog. However, neither soil temperature nor WT were found to be significantly related to the seasonal variation in CH_4 flux at the pasture, probably due to its low and highly variable

flux throughout the whole year. In addition, WT was found to exert a positive impact on CH_4 flux in warm period at the bog, suggesting that the rainfall during warm period is a crucial variable in regulating CH_4 flux. As a result, it is of particular significance to consider how future rainfall changes especially during warm growing season period to accurately predict the CH_4 flux in the future.

We found that the annual CH_4 flux at the bog was in the lower range of previously reported values for their counterparts elsewhere. This may be due to either the availability of elements or chemicals such as sulfate, ferric iron and nitrate (Kamal and Varma, 2008) which can serve as the competitive electrons acceptors for CH_4 production or low activity of methangenesis microbes. Detailed analysis of the biochemical elements and microbes (methanogens and methanotrophs) are needed to better understand the mechanisms. We found the agricultural management with drainage and grass cultivation lowered the CH_4 emission, in line with our hypothesis. This is not a novel finding as many previous studies had the similar finding, but such result from comparison study has the significance to accurately quantify the magnitude of the impact of agricultural management on the CH_4 flux in boreal peatlands.

Overall, the two chapters suggested different temporal patterns of CH_4 flux as well as underlying controls or mechanisms between undisturbed peatlands and managed ones, suggesting that the parameters and frameworks in modeling pristine peatlands are not suitable to those in the managed ones, and parameters special for modeling the CH_4 flux in managed peatlands should be developed.

In chapter 5, it determined the controls on the temporal dynamics of NEE and its components (GPP, ER) and compared the CO_2 flux between the bog and abandoned peatland pasture and the driving forces. Both the bog and abandoned peatland pasture showed pronounced and similar diel and seasonal patterns in CO_2 flux and so did the underlying drivers. GPP was more dependent on T_a than PPFD, and ER was determined more by T_1 than soil moisture at both sites. All these findings were similar to previous studies.

The abandoned peatland pasture was a stronger CO_2 sink, contrary to our hypothesis that the agriculturally managed peatlands should be hotspots of CO_2 source due to the high ER as suggested previously. This study suggested that agriculturally managed peatlands after long-term (more than 20 years) abandonment can function as a CO_2 sink, with the primary production of its plants outweighing the large CO_2 mineralization rates. This abandoned peatland pasture was drained and cultivated a high productive plant, RCG, which has been widely cultivated on the drained organic soil as bioenergy and mitigation of GHGs emissions in European countries. The finding in this paper gives some implication to estimate the CO_2 balance of agriculturally managed peatland, and land use intensity, time after abandonment, cultivated plant species should be considered when estimating or predicting the CO_2 balance of agriculturally managed peatlands.

The GPP in the abandoned peatland pasture was about 2.5 times that of bog but the difference cannot be explained by any environmental variables. Therefore, vegetation index data from the 16-day MODIS product (enhanced vegetation Index, EVI) was retrieved and the difference in the EVI (ΔEVI) between the two sites alone was found to explain the between-site GPP difference

by 65%. The ER at the abandoned peatland pasture was also significantly higher than that in bog and 45% and 52% of the difference can be explained by ΔEVI and ΔWT, respectively. Overall, our results suggested biotic variables should be important drivers in the inter-site difference of CO_2 flux and further analysis of them will help better understand the mechanisms. Analysis regarding to biomass, plant species composition, leaf area index, phenological indexes and photosynthetic capacity of key species should be conducted at high frequency to fully understand drivers governing the inter-site variation in CO_2 flux. However, we failed to know at what percentage of the larger ER in the abandoned peatland pasture was caused by its higher autotrophic respiration, which is linked with its higher aboveground biomass, or soil respiration due to more labile carbon and lower WT. Therefore, measurements of both soil respiration and total ER should be conducted at the same time to solve this issue.

In particular, we found both the GPP_{max} and light use efficiency of abandoned peatland pasture were higher than the values in the bog and ER at the abandoned peatland pasture (Q_{10} = 3.3) was more sensitive to soil temperature change than that at the bog (Q_{10} = 2.2). This suggested that agricultural management has changed the CO_2 cycling processes, making them totally different from those in the undisturbed peatlands. Therefore, the existing models to simulate GPP and ER based on the parameters from undisturbed peatlands may not be suitable for those managed ones, particular new models for managed peatlands are needed to estimate and predict the CO_2 flux at a large regional scale.

In the chapter 6, it compared the C balance and radiative forcing of all GHGs species (CO_2, CH_4 and N_2O) between the bog and abandoned peatland pasture. The C balance of each site and

difference between the two sites was mainly determined by CO_2 flux: the abandoned peatland pasture was a stronger C sink of -82 ~ -85 g C m^{-2} yr^{-1}, when compared to that of -14 ~ 4 g C m^{-2} yr^{-1} at the bog. For the C balance estimation, we only considered the vertical fluxes of CO_2 and CH_4 but ignored the lateral C loss as DOC, which may cause some overestimation of C sequestration capacity of both sites, especially in the abandoned peatland pasture with active drainage ditches

The radiative forcing of GHGs was mainly determined by CO_2 flux at the abandoned peatland pasture but by CH_4 flux at the bog. The abandoned peatland pasture showed stronger climate cooling function, when compared to the bog. N_2O is an important greenhouse gas but is always ignored in many previous studies when estimating the GHGs budget and radiative forcing of peatlands. This study considered the N_2O flux, in addition to the fluxes of CO_2 and CH_4 in both sites and suggested that N_2O flux was small and did not affect radiative forcing significantly, similar to findings in many previous studies. Neither significant seasonal variation in N_2O flux nor did any variables related to the variation were found at both sites. This study with very limited measurements only covered the growing season causes some uncertainty to estimate the cumulative N_2O flux. In addition, a few studies suggested high N_2O emission during wintertime with frequent soil freezing and thawing cycles, which contributed much to the annual flux. Therefore, high frequent measurements of N_2O flux throughout the whole year may improve the understanding of N_2O flux in our sites. Although we measured the DOC, TN contents in the peat pore water samples, the concentration of NO_3^-, NH_4^+, other elements, microbes which are closely related to N_2O flux are not known, preventing us from understanding the processes of N_2O flux in both sites. For the GWP/SGWP of GHGs estimation, the uncertainty in N_2O flux estimation

was the largest error source. Overall, our findings that agricultural management and abandonment promote the C and GHGs sequestration capacity of peatlands are novel, which are different from many previous studies.

Our findings emphasize that the C sequestration capacity and climate regulation function of managed peatlands can vary significantly between active managed peatlands and the abandoned ones. Previous studies of C and GHGs balance based on agriculturally managed peatlands under active use or even intensive use could overestimate the C loss and climate warming potential of those managed peatlands. Therefore, it is needed to distinguish the abandoned agriculturally peatands from those active managed peatlands to better estimate the role of managed peatlands in global C cycle and climate regulation and predict their response to future climate change.

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